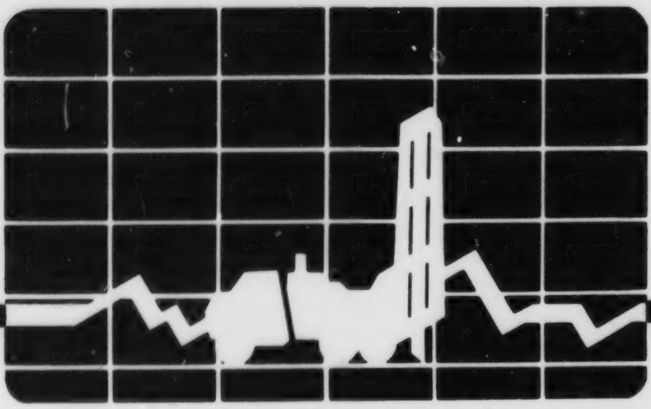
ConF-8105118--



CONF-8105118--

DE82 905817

Proceedings of the First National Ground-Water-Quality-Monitoring Symposium and Exposition

May 29-30, 1981
The Fawcett Center for Tomorrow
Columbus, Ohio

MASTER

DISTRIBUTION OF THIS DOCUMENT IS UNLIMITED

The First National Ground Water Quality Monitoring Conference and Exposition was jointly sponsored by the National Water Well Association and the U.S. Environmental Protection Agency.

NWWA Coordinating Committee
  David Nielsen
  Jay Lehr
  Kathy Butcher
  Bud Heiss
  Stu Smith
  Diana Kean

U.S. EPA Project Officer
  Jack Keeley, Chief
  Ground Water Research Branch
  U.S. Environmental Protection Agency
  Robert S. Kerr Environmental Research Lab
  Ada, Oklahoma 74820

Proceedings of the First National Ground-Water-Quality-Monitoring Symposium and Exposition.

Ground-water monitoring requirements of RCRA by Lawrence S Graves 2

A methodology for monitoring ground-water quality degradation by Richard M Tinlin 5

Problems of monitoring network design to detect unanticipated contamination by Hans-Olaf Pfannkuch 11

Monitoring in the vadose zone by L.G. Wilson 21

Monitoring of leachate migration in the unsaturated zone in the vicinity of sanitary landfills by Thomas M Johnson, Keros Cartwright and Rudolph M Schuller 31

A positive displacement ground-water sampling device by R W Gillham and P E Johnson 40

Recommended sampling procedures for monitoring wells by Rudolph M Schuller, James P Gibb and Robert A Griffin 43

Devising a ground-water monitoring strategy for a geologic repository for radioactive waste by Leo S Leonhart, Frank A DeLuca, N Thomas Sheahan and Lawrence M West 48

The radionuclide ground-water monitoring program for the Separations Area, Hanford Site, Washington State by Michael J Graham 54

Ground-water monitoring techniques for arid zone hazardous waste disposal sites by Robert F Kaufmann, Alan Gleason, R Brian Ellwood and Gerald C Lindsey 59

Cause and effect of cyclic changes in ground-water quality by Wayne Pettyjohn 67

Surface geophysical methods for ground-water monitoring by Hasan Yazicigil and Lyle V A Sendlein 40

Significance testing for impact evaluation by Frank A Rovers and Edward A McBean 86

U.S. Army installation restoration data management by Allen Shatto 91

***Papers presented by the following people were not available for publication at the time of printing of this volume:**
Edward E Everett/Henry A Baski/D Craig Shew/Franklin D Patton/David W Miller/
Richard R Parizek/W Scott Keys

DISTRIBUTION OF THIS DOCUMENT IS UNLIMITED

GROUND-WATER MONITORING REQUIREMENTS OF RCRA

By Lawrence S. Graves

Introduction

The May 19, 1980, promulgation of Resource Conservation and Recovery Act (RCRA) standards set into motion one of the largest regulatory programs ever enacted by the federal government. Protection of ground-water quality is a major objective of this program. Within the regulations is a multitude of permit deadlines and requirements for monitoring, recordkeeping, reporting and facility design and operation. This paper describes these monitoring requirements and presents them in an organized form that allows for easier comprehension.

The regulations are written such that an owner/operator of a hazardous waste facility may utilize one of three possible ground-water monitoring systems. The first system applied to a facility which *is not* assumed or known to be contaminating ground water. In most cases this system would be utilized at new facilities or facilities under construction.

The second system applies to a facility which *is* assumed or known to be contaminating ground water. This alternate system would be oriented toward defining the extent of contamination and monitoring its further migration.

The third system applies to a facility where a justification can be provided for a lesser degree of monitoring because the owner/operator can demonstrate a low potential for migration of hazardous waste constituents from the facility via the uppermost aquifer to water-supply wells or surface water.

A series of flowcharts has been developed that depict the important dates, sequence of events and installation requirements of the ground-water monitoring systems (Figures 1 through 3). Each flowchart is followed by a detailed description of the separate components in the flow chart. All references given in this paper are contained in the "EPA Hazardous Waste and Consolidated Permit Regulations" in the May 19, 1980, issue of the *Federal Register*.

The regulations do not specifically address the need for remedial actions. It may be presumed that the need for these measures at sites with proven contamination will be based, in part, on the EPA's performance standards and on the potential for future contamination of water-supply wells or surface waters near the individual site in question. Facilities that were inactive or abandoned prior to November 19, 1980, are not subject to these regulations.

First Monitoring System Alternative

The first ground-water monitoring system requirements can be divided into:

(1) the deadlines and requirements applicable to the installation of the ground-water monitoring system (Figure 1), and

(2) the required procedures to be followed in the event water-quality samples indicate that ground water is being degraded (Figure 2).

Figure 1 details are as follows:

Block 1

By November 19, 1981, which is one year after the effective date of these regulations, the owner/operator is required to implement the ground-water monitoring program, including the installation of monitoring wells required in Sec. 265.91 (Ground-water monitoring systems, page 33240). As stated in the regulations, monitoring wells must be installed hydraulically upgradient from the limit of the waste management area. The number, locations and depths of the wells must be sufficient to yield ground-water samples that are representative of background water quality in the uppermost aquifer near the facility and not affected by the facility. In addition, monitoring wells must be installed hydraulically downgradient at the limit of the waste management area. Their number, locations and depths must ensure that they immediately detect any "wastes" that migrate from the waste management area to the uppermost aquifer.

In addition to installing the system, the owner/operator is required to prepare a ground-water quality assessment-plan outline. The outline represents (in a preliminary scheme) the procedures that will be followed to assess the extent to which wastes have entered ground water in the event water-quality results indicate a statistically significant difference between present and

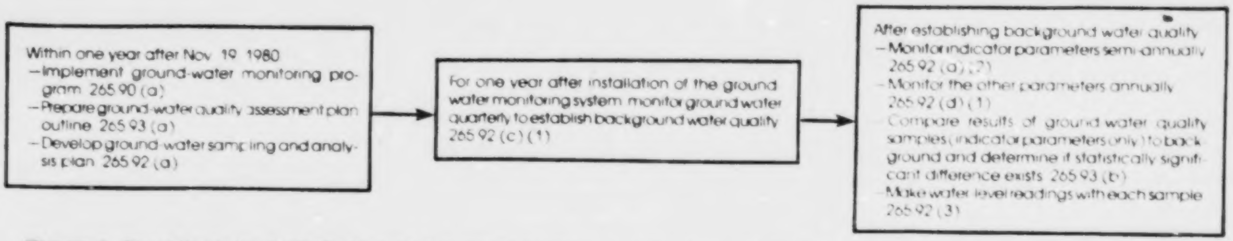

Within one year after Nov 19 1980
—Implement ground-water monitoring program 265.90 (a)
—Prepare ground-water quality assessment plan outline 265.93 (a)
—Develop ground-water sampling and analysis plan 265.92 (a)
For one year after installation of the ground water monitoring system monitor ground water quarterly to establish background water quality 265.92 (c) (1)
After establishing background water quality
—Monitor indicator parameters semi-annually 265.92 (a) (2)
—Monitor the other parameters annually 265.92 (d) (1)
—Compare results of ground-water quality samples (indicator parameters only) to background and determine if statistically significant difference exists 265.93 (b)
—Make water level readings with each sample 265.92 (3)

Figure 1. Deadlines for installation of the ground-water monitoring system and requirements for routine sampling applicable to new facilities and existing facilities not assumed (or known) to be affecting ground water.

background water quality.

Finally, the owner/operator must develop and have on file a ground-water sampling and analysis plan. This plan must include procedures and techniques for sample collection, sample preservation and shipment, analytical procedures and chain-of-custody control.

Block 2

For a period of one year after the owner/operator has installed the ground-water monitoring system, he must monitor ground water in all wells to establish background water quality. He must take samples every three months and analyze for the following indicator parameters:

(1) parameters characterizing the suitability of the ground water as a drinking water supply: arsenic, barium, cadmium, chromium, fluoride, lead, mercury, nitrate, selenium, silver, endrin, lindane, methoxychlor, toxaphene, 2,4-D, 2,4,5-TP silvex, radium, gross alpha, gross beta, turbidity and coliform bacteria;

(2) parameters establishing ground-water quality: chloride, iron, manganese, phenols, sodium, sulfate and

(3) parameters used as indicators of ground-water contamination: pH, specific conductance, total organic carbon and total organic halogens.

Replicate measurements are required for the indicator parameters analyzed from samples taken from the upgradient wells. The initial background arithmetic mean and variance must be determined by pooling the replicate measurements of the respective parameter concentrations or values.

Block 3

After the first year, all monitoring wells must be sampled and the samples analyzed with the following frequencies:

- Parameters used to establish ground-water quality are sampled and analyzed annually.
- Parameters used as indicators of ground-water contamination are sampled and analyzed semi-annually.

The owner/operator must compare the results of the indicator parameter analyses with the computed background and determine, by use of the student's t-test, if a significant difference exists (at the 0.1 confidence interval). In addition, water-level readings must be taken to determine if the hydraulic gradient in the area has changed.

If a statistically significant difference in water quality occurs between current and background water quality, the sequence of events which the owner/operator must follow is depicted in Figure 2 and detailed below.

Block 1

Resample, split the sample in two, and analyze to confirm that the difference is not a result of laboratory error.

Blocks 2 and 3

If the difference is verified by the resample, the RA (Regional Administrator) must be notified within seven days. However, if the difference proves to be the result of laboratory error, the owner/operator returns to routine monitoring.

Block 4

Within fifteen days after notifying the RA, the owner/operator is required to develop a ground-water quality assessment plan. This plan should be a more detailed expression of the outline prepared during the first year after November 19, 1980 (Figure 1, Block 1). The plan must be certified by a qualified geologist or geotechnical engineer and must specify:

(1) the number, location and depth of wells;

(2) sampling and analytical methods for those hazardous wastes or hazardous waste constituents in the facility;

(3) evaluation procedures, including any use of previously gathered ground-water quality information

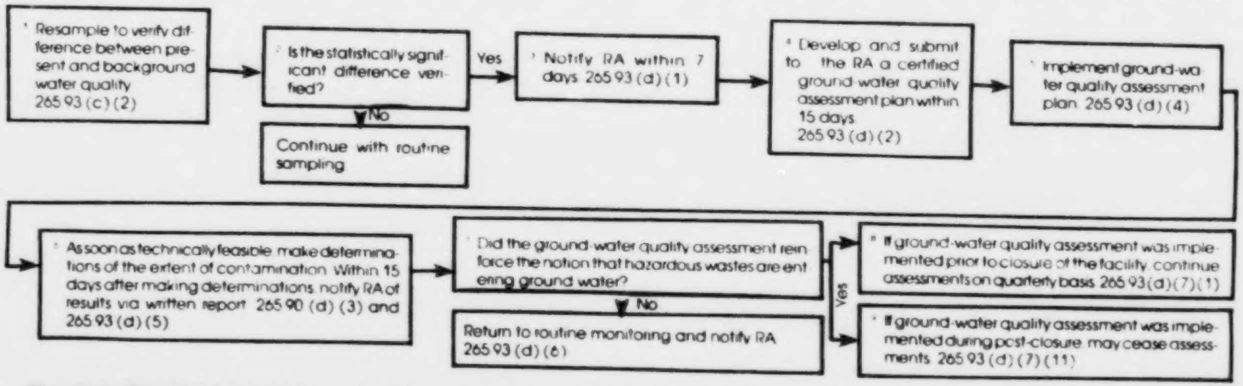

Resample to verify difference between present and background water quality 265.93 (c) (2)
Is the statistically significant difference verified?
Yes
No
Continue with routine sampling
Notify RA within 7 days 265.93 (d) (1)
Develop and submit to the RA a certified ground-water quality assessment plan within 15 days 265.93 (d) (2)
Implement ground-water quality assessment plan 265.93 (d) (4)
As soon as technically feasible make determinations of the extent of contamination. Within 15 days after making determinations, notify RA of results via written report 265.90 (d) (3) and 265.93 (d) (5)
Did the ground-water quality assessment reinforce the notion that hazardous wastes are entering ground water?
No
Return to routine monitoring and notify RA 265.93 (d) (6)
Yes
If ground-water quality assessment was implemented prior to closure of the facility, continue assessments on quarterly basis 265.93 (d) (7) (1)
If ground-water quality assessment was implemented during post-closure, may cease assessments 265.93 (d) (7) (11)

Figure 2. Required procedures in the event water-quality samples indicate that hazardous wastes are entering ground water.

and

(4) a schedule of implementation of the plan.

Block 5

The owner/operator must implement the plan and initiate the determination of:

(1) the rate and extent of migration of the hazardous waste or hazardous waste constituents in the ground water and

(2) the concentrations of the hazardous waste or hazardous waste constituents in the ground water.

Block 6

As soon as it is technically feasible, the owner/operator must make the determinations specified in Block 5. Within 15 days after making these determinations, the owner/operator must submit to the RA a written report containing an assessment of the ground-water quality.

Block 7

If the assessment demonstrates that hazardous wastes are present in ground water, proceed with activities specified in Blocks 8 and 9. However, if the assessment demonstrates that hazardous wastes are not entering ground water, the owner/operator notifies the RA of the results and returns to routine monitoring.

Blocks 8 and 9

If the ground-water quality assessment verifies the existence of a contamination problem and if the assessment began before the facility was closed, the owner/operator must reassess the contamination problem every three months until final closure of the facility. After closure, the assessments may cease. If the assessment began during the post-closure period, the assessment may cease.

Second Monitoring System Alternative

Subpart F has been written such that if an owner/operator of a hazardous waste facility assumes or knows that his facility is impacting ground water, he may elect to install an alternate ground-water monitoring system, that is, a system different from the one required under 265.91 and 265.92. Figure 3 illustrates the sequence of events necessary to implement the alternate system.

Block 1

Within one year after November 19, 1980, the owner/operator is required to submit a ground-water quality assessment plan that is certified by a qualified geologist or geotechnical engineer, which specifies:

(1) the number, location and depth of wells;

(2) sampling and analytical methods for those hazardous wastes or hazardous waste constituents in the facility;

(3) evaluation procedures, including any use of previously gathered ground-water quality information and

(4) a schedule of implementation of the plan.

The owner/operator must implement the plan and initiate the determination of the rate and extent of migration and the concentrations of the hazardous waste or hazardous waste constituents in the ground water.

Block 2

As soon as it is technically feasible, the owner/operator must make the determinations specified in Block 1. Within 15 days after making these determinations, the owner/operator must submit to the RA a written report containing an assessment of the ground-water quality.

Block 3

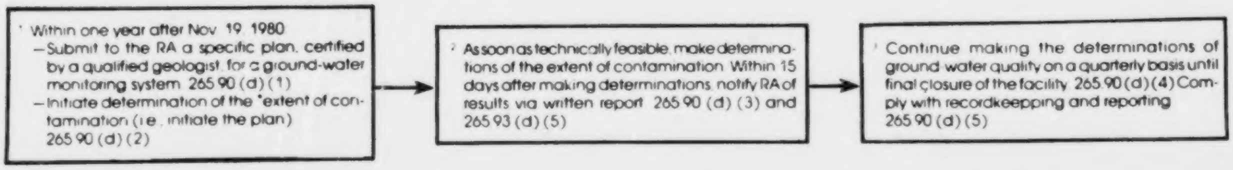
The owner/operator is required to reassess the rate and extent of migration and the concentrations of the hazardous waste and hazardous waste constituents in the ground water every three months until the facility is closed.

Third Monitoring System Alternative

The owner/operator of a facility having a low potential for migration of contaminants may install a system that provides a lesser degree of monitoring. This particular system is not illustrated by a flowchart because the nature of the monitoring system will be quite varied. The system could consist of less monitoring, less frequent sampling, analysis of fewer chemical parameters or all of the above. The only requirement is that the demonstration of a low potential for migration of contaminants must be certified by a qualified geologist or geotechnical engineer and must be in writing and kept on file at the facility.

Biographical Sketch

***Lawrence S. Graves**, a senior hydrologist with the ground-water consulting firm of Geraghty & Miller in Annapolis, Maryland, earned his M.S. in hydrogeology from Bowling Green State University in 1977. He has worked for the U.S. EPA and was involved in drafting the ground-water regulations of the Resource Conservation and Recovery Act.*



1 Within one year after Nov. 19, 1980
- Submit to the RA a specific plan, certified by a qualified geologist, for a ground-water monitoring system. 265.90 (d) (1)
- Initiate determination of the *extent of contamination (i.e., initiate the plan) 265.90 (d) (2)

2 As soon as technically feasible, make determinations of the extent of contamination. Within 15 days after making determinations, notify RA of results via written report. 265.90 (d) (3) and 265.93 (d) (5)

3 Continue making the determinations of ground water quality on a quarterly basis until final closure of the facility. 265.90 (d) (4) Comply with recordkeeping and reporting 265.90 (d) (5)

*For the purpose of brevity, the term "extent of contamination" is used instead of the more exact regulatory language.

Figure 3. Deadlines for installation of the alternate ground-water monitoring system and requirements for ground-water quality assessments applicable to facilities that are assumed (or known) to be impacting ground water.

A METHODOLOGY FOR MONITORING GROUND-WATER QUALITY DEGRADATION

By Richard M. Tinlin

Introduction

A frequent comment regarding the use of methodologies to assess ground-water quality degradation is that they fall apart when applied at the site-specific level. The basis for this opposition is a conviction that every waste disposal site is fundamentally a unique hydrogeologic unit.

Webster's Dictionary defines the concept of a methodology as "the science which describes and evaluates arrangements of materials of instruction." One of the most well-known methodologies is the "scientific method" taught in all basic science courses. The scientific method is general in scope but is readily applied to specific investigations. Methodologies are particularly useful in helping to avoid the pitfalls and mistakes encountered by our predecessors. They can also be invaluable tools for transferring knowledge from the experienced to the inexperienced.

A number of methodologies for monitoring ground-water quality presently exist or are under development. It is incorrect to assume that there is or will be only "one" methodology that should win out over all the others. Some are rigidly tailored to a specific situation and others are quite general. Depending on one's perception of ground-water monitoring, the great variety of methodologies can result in confusion and concern. This comes about because the goals and objectives vary greatly within the overall concept of ground-water quality monitoring and because each individual involved is biased depending on his relative position within the system. For example, a state level administrator typically is faced with the dilemma of dealing with the large number of problems brought to his attention from all over his state and then allocating monitoring resources to those sources perceived to present the greatest threat. To accomplish this he needs information in a form that can be readily understood and presented to the various funding agencies, federal, state and private. An individual working at the site-specific level will be concerned primarily with his immediate site-specific objectives. He will do his best to convince his supervisors that he needs funding.

The EPA Ground-water Protection Strategy

The EPA ground-water protection strategy presents a number of national goals, objectives and general methods along with a timetable for meeting these goals and objectives. The "strategy" explains the position of the EPA at the federal level but leaves it open to the states, except for a few general directives, to develop their own "ground-water protection strategies" within the framework of the federal and state regulations.

Major difficulties exist in trying to formulate a strategy at the national level that can be applied cost-effectively all the way down to the site-specific level. It is difficult to argue with the general goals of the EPA strategy, i.e., to assess, protect and enhance the quality of ground waters. The management approach proposed in the EPA strategy, however, is subject to considerable debate. The concept of a ground water classification scheme is particularly arguable. Classification of the quality of the nation's ground waters to the degree needed will be a monumental and potentially endless task. Experienced hydrologists know that even at the site-specific level, ground-water quality varies significantly in both the vertical and lateral dimensions. The U.S. Geological Survey, along with other federal and state agencies, has spent many years trying to classify the nation's ground waters. They have made some progress but a great deal of time and money has been required and the job is far from complete. The economics of any attempt to classify the nation's ground waters should be carefully evaluated prior to implementation, as the cost will prove to be the weak link. The methodology, perhaps better described as a "framework," presented in this paper is a more logical approach to the problem as it attacks the problem on a priority basis. It is source-oriented and concentrates on the identification of priority sources and areas so that monitoring resources can be allocated in the most effective way.

The Tempo Ground-water Monitoring Methodology

In 1976, the General Electric Co.'s Center for Advanced Studies completed the development of a 15-step methodology for monitoring the effect of man's activities on ground-water quality (Todd, et al.

1976). The basic purpose of the methodology was to provide a framework for the planning and development of a monitoring program. The proposed steps of the methodology are listed below:

Step 1—Select the area for monitoring;
Step 2—Identify pollution sources, causes and methods of disposal;
Step 3—Identify potential pollutants;
Step 4—Define ground-water use;
Step 5—Define hydrogeologic situation;
Step 6—Describe existing ground-water quality;
Step 7—Evaluate infiltration potential of wastes at the land surface;
Step 8—Evaluate mobility of pollutants from the land surface;
Step 9—Evaluate attenuation of pollutants in the saturated zone;
Step 10—Prioritize sources and causes;
Step 11—Evaluate existing monitoring programs;
Step 12—Identify alternative monitoring approaches;
Step 13—Select and implement the monitoring program;
Step 14—Review and interpret monitoring results; and
Step 15—Summarize and transmit monitoring information.

A great deal of discussion and compromise went into the ordering of the steps of the methodology. A basic premise was that the goal of the methodology was to define or identify ground-water pollution resulting from man's activities and not from natural sources. It was envisioned that the methodology would be applied at the substate level and in a serial manner for a specific monitoring area. Under a subsequent contract between General Electric and the EPA, an attempt was made to apply the methodology to energy-related development in the Western United States. The two types of development selected for study were coal strip mining and oil shale extraction. As the project progressed it gradually became apparent that the methodology needed to be applied in an iterative manner when dealing with a multi-source area. It also became obvious that the goals and objectives of the steps are significantly different, ranging from those at the administrative, regulatory level to those at the site-specific level. To accommodate these needs a modified version of the methodology or framework for monitoring ground-water quality is proposed below.

A Methodology for Monitoring Ground-water Quality

The methodology presented below is multi-leveled and phased (see Tables 1, 2 and 3). It is multileveled to encompass the differing goals and objectives which exist at the state, local and site- or source-specific levels. Each level consists of one or more tasks or steps. At the source-specific level, five steps or objectives are proposed for evaluating each source. Which step or steps are implemented will depend on the state in which the ground-water contamination is found to exist, i.e., potential, present or past.

For the purposes of this article, a potential source is defined as one which has virtually no monitoring underway but because of the characteristics of the materials disposed of in the source, the method of disposal and the site hydrogeology, a significant potential for ground-water contamination exists. Present pollution is defined as the condition where ground water contamination can be linked to an existing source. Past pollution is defined as the condition where the source or a party responsible for the source can no longer be identified.

The methodology is phased to accommodate the fact that monitoring programs are dynamic. As a monitoring program develops it must be revised to meet the needs of changing objectives and new information. Three phases are proposed for the methodology. The primary objectives of the first phase are:

- to divide up a state based on administrative, physiographic and priority considerations;
- to identify and make a preliminary prioritization of those sources in each area thought to need immediate attention;
- to examine those sources showing the greatest potential to contaminate ground water and determine what data and information already exists, what data and information are needed and what it will cost to obtain the needed information. The existing data will be used to upgrade the source prioritization list first at the area-wide level and then at the state level.

The identification of the existing data and information gaps in Phase 1 will allow the design of a more comprehensive monitoring program for implementation in Phase 2 of the methodology. The primary objective of Phase 2 will be the design and implementation of new monitoring to fill in the data gaps and to provide the necessary information for updating of the source priority listing. A secondary goal will be to provide information on the cost of the additional monitoring.

The primary objectives of the third phase of the methodology are to monitor the effectiveness of any controls or abatement procedures imposed on a source and to revise the monitoring activities as needed. A "sunset" provision, to phase out or end monitoring programs that are no longer needed, should be provided.

The following discussion of the methodology assumes that it is being applied in a state where no overall statewide monitoring programs exists. At the source-specific level the special case situation where no monitoring is underway is assumed.

Phase 1—State Administrative Level

A basic assumption for the following discussion is that a state administrator has been or will be appointed. He will serve as the overall coordinator for this program. He will also interface with the EPA Ground-Water Protection Strategy and other federal programs which impact on his program. Table 1 offers an overview of Phase 1. The administrator's first task will be to examine the state's political and institutional structure. This information will be essential in dividing up the state in a realistic and effective manner. It will be used to assign responsibility for the different classes of potential ground-water contamination to the appropriate agencies. This does not necessarily mean that these agencies will always actually do the monitoring, but it does mean that monitoring will be carried out where necessary.

The second task will be to look at the physiography of the state and work out a compromise between the state's administrative structure and its physiography so that the state can be divided up into "Designated Ground-Water Quality Management Areas" (DGWQMA'S). An individual or agency in each DGWQMA will need to be identified to serve as the interface between the state administrator and the field activities in his area.

A third task will be to develop a priority ranking mechanism so that as pollution sources are discovered and evaluated in each DGWQMA, some means will exist for allocating monitoring resources to those sources perceived to present the greatest threat.

Phase 1—Field Administrative Level

Once the state administrative level is set up, a first action of the state administrator will be to direct the managers of each DGWQMA to carry out the first task at the field administrative level, an inventory of potential sources of pollution and methods of waste disposal. The degree of detail of this inventory will depend on available funding, manpower, the size and complexity of the area and the types of pollution sources present in the area. At periodic intervals, each director will report the results of his inventory to the state level. This will be a continuing process—as new sources develop they will be entered onto the list. The first inventory will identify the obvious present sources. Some sources falling into the "past" category will be more difficult to locate and will require some "detective" work.

Phase 1—Source-Specific Level

Those sources which appear to present a significant potential threat to ground-water quality will need to be monitored in greater detail. Five steps are proposed for evaluating a potential source. Each step has a specific purpose and the steps are arranged in a logical sequence related to the order in which ground-water contamination generally takes place. Depending on a particular method of waste disposal (e.g. injection through wells), some steps will be bypassed. The overall goal of each of the five steps in Phase 1 at the source-specific level will be to examine existing monitoring and to determine what data and monitoring gaps exist. Any existing data, if adequate, will be interpreted and used to update the priority ranking of the source. The data and monitoring gaps identified will be used to develop a more comprehensive monitoring program for implementation in Phase 2 of the methodology.

Step 1-Identify Potential Pollutants

The purpose of this step is to characterize a pollution source in terms of the types, concentrations and amounts of potential contaminants present. This is an essential step in ascertaining which monitoring method will need to be implemented in Phase 2 of the methodology. Many monitoring programs are conducted for a period of time before it is realized that an important contaminant has been inadvertently excluded from the monitoring effort. In many instances it is not possible to make up for the information lost, i.e., when it is desired to determine how a contaminant is attenuated over time.

The information gained from this step alone may be sufficient to relegate the pollution source to a very low standing in terms of priority for monitoring, when compared against other sources. Any existing information useful in making a preliminary priority judgment will be recorded in Step 6a, in a form readily useable for making the comparison with other sources. This will allow those sources of significance to be moved up through the ranking system to a position consistent with their importance as a potential source of ground water contamination. Any data and monitoring gaps identified will be used to design a more comprehensive monitoring program for implementation in Phase 2.

Step 2-Identify Ground-water Use

Some information on ground-water use and quality will almost always be available near a potential source of contamination. This information can be used to make an initial appraisal of the background water quality and a judgment on the potential threat that contaminants identified in a source pose to

Table 1 — Monitoring Methodology
Phase 1

Level	
State Administrative Level (SAL)	Task 1. Administrative considerations Task 2. Physiographic considerations Task 3. Priority considerations
Field Administrative Level (FAL) for Designated Ground-water Quality Monitoring Area (DGWQMA)	Task 1. Inventory potential pollution sources and methods of disposal
Site- or Source-Specific Level (SSL)	Evaluate existing monitoring and determine data gaps
	Step 1. Identify potential pollutants Step 2. Identify ground-water usage Step 3. Evaluate infiltration potential Step 4. Evaluate pollutant mobility in vadose zone Step 5. Evaluate pollutant mobility in saturated zone Step 6. Prioritize source a. list potential pollutants, amounts and concentrations b. ground-water usage c. amounts infiltrating d. mobility of infiltrating pollutants in the vadose zone e. mobility of pollutants reaching the saturated zone

Table 2 — Monitoring Methodology
Phase 2

Level

SAL Institute any administrative changes required and upgrade overall potential pollution source prioritization

Task 1. *
Task 2. *
Task 3. *

FAL Task 1. Update potential pollution source inventory

Design and implement new monitoring

SSL	Specific Monitoring Objective	Sampling Method	Cost	Frequency	Cost	Analysis	Cost	Total Cost
Step 1.	*	Existing monitoring						
	*	Monitoring gaps						
Step 2.	*	Existing monitoring						
	*	Monitoring gaps						
Step 3.	*	Existing monitoring						
	*	Monitoring gaps						
Step 4.	*	Existing monitoring						
	*	Monitoring gaps						
Step 5.	*	Existing monitoring						
	*	Monitoring gaps						

Step 6. Upgrade prioritization based on new information

a. *
b. *
c. *
d. *
e. *

* Task and step descriptions remain the same as those in Table 1.

ground-water users. This information will be recorded in Step 6b. This methodology does not advocate extensive background monitoring, because the expense of such an effort is not justified until it is demonstrated that a significant threat exists. The combination of potential contaminants present in a source and the use of the ground water that is threatened by the contaminants can be used to upgrade or downgrade a particular pollution source when compared against other sources.

The next three steps constitute an evaluation of the hydrogeologic framework. They are ordered so that if a potential contaminant fails to infiltrate or is attenuated in either the unsaturated zone or the saturated zone, the monitoring effort can be scaled down or otherwise altered to avoid the unnecessary expense of installing a large number of monitoring wells.

Step 3-Identify Infiltration Potential

For most land surface waste disposal methods, whether the material is allowed to infiltrate or not will be an important consideration in justifying the expense of additional monitoring and in upgrading the source as a potential threat to ground water. A variety of methods are available, including the water budget method for determining infiltration rates. The costs for implementing these methods vary significantly and judgments on which method is most suitable will depend on source-specific considerations. Existing information will be used to estimate the importance of the source as a potential source of ground-water contamination and recorded in Step 6c. Those monitoring and data gaps identified in the process of examining the existing data will be used as the basis for designing and selecting the methods to be implemented in Phase 2, Step 3 of a program.

Step 4-Evaluate Pollutant Mobility in the Vadose Zone

Potential contaminants moving through the vadose zone are subject to a variety of biological, chemical and physical mechanisms which can reduce the threat of a potential contaminant significantly or, in some cases, entirely. These attenuating mechanisms are of great eco-

Table 3 — Monitoring Methodology
Phase 3

Level

SAL Continue phase 2 revisions where needed
Task 1. *
Task 2. *
Task 3. *

FAL Task 1. Revise potential pollution source inventory

Operate and revise monitoring program in response to controls

SSL Monitoring changes needed to assess effectiveness of controls

	Sampling Method	Cost	Frequency	Cost	Analysis	Cost	Total Cost
Step 1.	*						
Step 2.	*						
Step 3.	*						
Step 4.	*						
Step 5.	*						

Step 6. Revise prioritization based on effectiveness of control measures
a. *
b. *
c. *
d. *
e. *

* Task and step description remain the same as those in Table 1.

nomic importance. It is not unusual to encounter situations where monitoring wells have been completed in the saturated zone at great expense to monitor potential contamination that will never arrive. These wells are usually justified in terms of obtaining background information when in reality it is not needed.

Any existing useful information found to exist will be interpreted and used to adjust the priority ranking for the source. It will be recorded in Step 6d. The identified data and monitoring gaps will be used to aid in the design of the monitoring effort to be implemented in Phase 2, Step 4.

Step 5-Evaluate Pollutant Mobility in the Saturated Zone

No matter when ground-water contamination actually takes place, once ground water becomes contaminated it is difficult and expensive to resolve the problem. It would seem logical to assume that the point in time when a contaminant passes from the vadose zone into the saturated zone is the point at which ground water becomes contaminated. However, when certain contaminants interact with saturated conditions, they are readily attenuated and thus no longer a potential threat to anything except a well in very close proximity.

As in the previous steps the important objectives for this step will be to identify any useful existing information and to identify monitoring gaps and additional data requirements. Any existing information of value will be used to upgrade the priority file in Step 6e. The information gathered on data and monitoring gaps will be used to assist in the design of the monitoring program for Phase 2, Step 5.

Step 6-Prioritize Source

On the first pass through the preceeding five steps, some information will be turned up for each step and filed in this step for easy interpretation. This information may be useful in making a preliminary ranking of the source so that monitoring resources can be properly directed. Any source rated to be a significant threat to ground-water quality will be elevated to the field administrative level priority file and, if it survives there, it may be further elevated to the state administrative level priority file as a priority source. It is envisioned that, because funds are always limited, only those sources having the highest potential to degrade ground water quality will be selected for monitoring. Not all funding will come from the public sector. However, because of regulatory requirements, it will still be in the best interests of both the public and private sectors to allocate their monitoring resources effectively.

Phase 2—State Administrative Level

As a state's ground-water quality

monitoring program develops, it will be necessary from time to time to make administrative and institutional changes in the program at the state level. It is anticipated that as information on the distribution and severity of the various types of potential sources of ground-water contamination is generated, some areas of a state will be selected for more attention than others.

An ongoing activity at the state level will be to constantly upgrade the potential pollution source priority list (see Table 2). The information supporting the position of a source on the priority list should be reduced to a simple and graphic format, for use in convincing the various funding agencies to allocate resources to match the threat. Massive state-level data storage programs should not be attempted. Matching a particular threat to ground-water quality may involve funding a monitoring program, implementing controls on the source or area of contamination and maintaining the monitoring program to determine if the controls are working.

Phase 2—Field Administrative Level

A continuing task at the field administrative level will be to update the potential pollution source inventory. It is envisioned that sources of ground-water contamination will continue to surface for some time in all three of the major categories—potential, present and past. In time, all or almost all of the sources falling in the present and past categories will be identified. Those falling in the potential category can be identified if a screening mechanism is incorporated into local land-use planning procedures to identify potential sources. This way it will not be necessary to devote resources to classifying an area's ground water until a need is demonstrated. Spills resulting from accidents are a special case of potential contamination, but once identified the methodology would be implemented as it stands.

Phase 2—Source-Specific Level

Using the information on existing monitoring and data and monitoring gaps generated in the Phase 1 source-specific review, a monitoring program would be designed for the particular source under investigation (see Table 2). The procedure would begin with a listing of the specific monitoring objectives for each of the first five steps. Then for each objective, the method or methods most suitable for meeting the objective—taking into consideration site-specific factors along with particulars on costs, recommended sampling frequencies, methods of analyses and costs for the latter two items—would be identified.

A major advantage of planning and pricing a monitoring program in advance is that it allows planners to see what they are up against. Adjustments almost always have to be made in a program. Many programs suffer an early death because the whole budget is spent before they get off the ground. Several iterations may be needed between the funding agency and site operator to arrive at a realistic monitoring effort.

Once a program has been given the go-ahead, it should be subject to constant review regarding its necessity. The five steps proposed have been designed so that if at any time the potential of the source to contaminate ground water can be shown to be reduced significantly, the emphasis on the program can be adjusted correspondingly. This choice should not always be left up to the site operator, depending on his vested interest in keeping the program going.

The output of the five steps would be interpreted and information supporting its rank as a potential source of ground-water contamination would be stored in the appropriate places in Step 6, the priority file. The information in this file would in turn be taken to the field administrative level for comparison against other sources on an area-wide basis. Here it would compete for funds. If it survives it may make it to the state-level list.

Phase 3—State Administrative Level

Sources advancing to Phase 3 would be those requiring various control measures to prevent or abate ground-water contamination (see Table 3). The state administration would locate funding to support the implementation of the controls or provide the regulatory impetus needed to encourage private interests to implement appropriate controls.

An important function of the state administrator's office would be to review those sources for which controls have been implemented, make determinations on how effective the controls are and make any necessary adjustments in the programs.

Phase 3—Field Administrative Level

The field-level administrators would carry out activities similar to those proposed above for the state level administrator. His office should serve as a buffer between the state and source-specific levels, intercepting and handling those problems within its ability. The field administrator would continue to upgrade the potential pollution source inventory and upgrade the priority source list.

Phase 3—Source-Specific Level

At the source-specific level, the basic goal will be to maintain and adjust the monitoring program in response to the effectiveness of the controls imposed. If the controls are working, the source priority may be lowered. This would require a procedure to notify the field- and state-level administrators so that they can reallocate funding effectively.

References

Todd, D.K., Tinlin, R.M., Schmidt, K.D. and Everett, L.G. Monitoring Ground-water Quality: Monitoring Methodology, U.S. Environmental Protection Agency, Las Vegas, Nevada, 1976.

Biography

Richard A. Tinlin is an independent consulting hydrogeologist based in Camp Verde, Arizona.

PROBLEMS OF MONITORING NETWORK DESIGN TO DETECT UNANTICIPATED CONTAMINATION

by Hans-Olaf Pfannkuch

Special Problems of Unanticipated Contaminant Release

The unanticipated release and propagation of contaminants in the ground-water flow system is a special case of pollution. Strategies to handle such a contamination case are different from those employed in more current situations where the likelihood of pollution could be foreseen. This article deals with two problems of this kind.

The first problem is the accidental occurrence of spills at a point along an extended, linear route such as a pipeline break or a tank car accident. An unforeseeable element of chance or randomness is introduced because of the unknown location and time of the incident. The second problem involves leachate generation and propagation from unregulated or pre-regulation abandoned land disposal operations. The random element in this case is that, in general, the location, nature and strength of the source are unknown when the problem first surfaces. Both situations are characterized by the lack of general information about the hydrogeologic characteristics of the site and the dynamics of the ground-water flow system near the source. It is economically unfeasible to gather detailed information for every foot of transmission route before the actual occurrence of an accident, nor is it feasible to expend large efforts for studies on hundreds or thousands of suspected disposal sites before a problem is detected. This means that, in a situation in which a rapid response is necessary, the degree of preparedness is much lower than for those cases where potential contamination from a known source of fixed location has been planned for in the design.

The basic commonality in managing accidental spills along extended routes and contaminant detection from unknown sources is that rapid countermeasures have to be taken and that monitoring operations are tightly interwoven with the basic information gathering process, the development of the observation network and schedule, and the direction of the abatement and cleanup activities. Especially in the early stages of development, all of these operations occur simultaneously. This article discusses specific monitoring strategies to meet these special situations.

Monitoring in the General Pollution Control Context

In order to optimize the design and operation of monitoring systems it is appropriate to take a holistic view of monitoring as part of the information component in the overall management of pollution control operations (Pfannkuch, 1975). Controlling a spill or contamination case is a decision and management process with goals and objectives, economic and technological constraints and with optimization procedures and decisions about alternative courses of action. Management goals in response to a pollution incident may range from total removal of the pollutant and reconstitution of the environment to pre-pollution levels over abatement and mitigation to simply minimizing and containing the damage. Monitoring operations are a tool to assess the situation, to guide the action, and to evaluate its efficiency and compliance with set standards.

Figure 1 shows the relation of monitoring operations and monitoring systems design to the main management objectives, but also to the constraints imposed by the physical and technological realities, and by the socioeconomic environment (Pfannkuch and Labno, 1976). The economic optimization of the monitoring process is a resource allocation problem with the goal of developing the most cost effective monitoring program in order to obtain the maximum protection of the ground-water resource from a technological point of view (Todd et al. 1976).

This article concentrates on the technological aspects of optimization, because before an economic decision can be reached a technological background has to be created on the basis of which different alternatives can be evaluated. The first important element in designing an efficient monitoring system is the proper understanding of the mechanics and dynamics of contaminant propagation, i.e., whether the contaminant moves as a soluble phase or by multiphase flow, whether the controlling mechanisms are described by vadose or saturated flow, piston displacement or diffusion-dispersion among other things. In addition, transport parameters of the flow system, such as permeability, porosity, capillarity and dispersivity of the material through which flow occurs must be evaluated. Along with the macroscopic or extensive parameters, such as the boundaries of the aquifer system and its hydrodynamic variables potential,

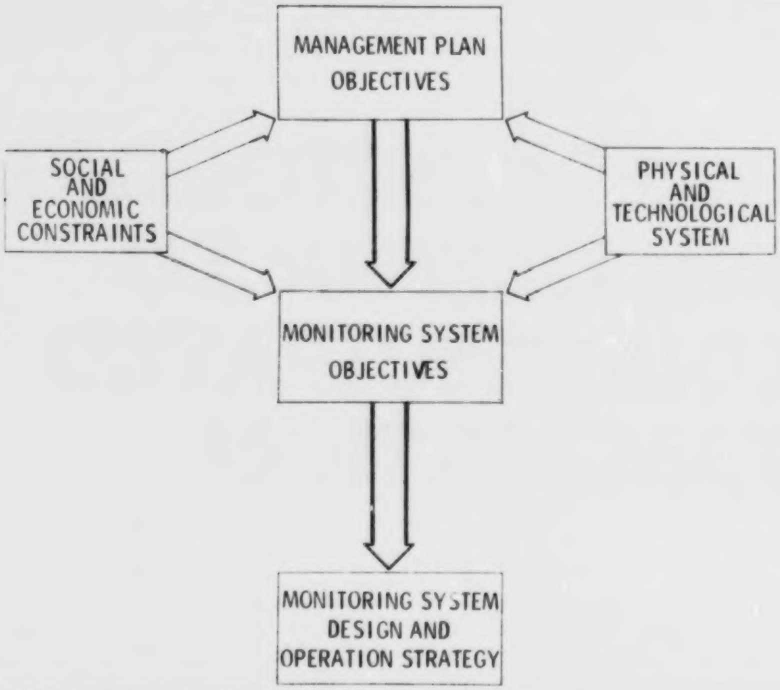

MANAGEMENT PLAN OBJECTIVES
SOCIAL AND ECONOMIC CONSTRAINTS
PHYSICAL AND TECHNOLOGICAL SYSTEM
MONITORING SYSTEM OBJECTIVES
MONITORING SYSTEM DESIGN AND OPERATION STRATEGY

Figure 1 Monitoring system design and operation.

			RELATION TO ENVIRONMENT		
	SOURCE	OPERATION	OPEN	RESTRICTED	ISOLATED
1	WASTE	DISPOSAL	☒		■
2	BY-PRODUCT	LAND-USE PROCESS			
3	PRODUCT	STORAGE TRANSPORT DISTRIBUTION			
	ANTICIPATED RELEASE		ACCEPTED	CONTROLLED	ACCIDENTAL

Figure 2 General contaminant release matrix.

water table configuration and potential gradients. these factors determine the direction and velocity of movement and dispersion of the contaminant. Based on this information, the location and number of monitoring wells, and the frequency of sampling can be established. The reconnaissance and information gathering operations in this first phase are not really monitoring operations per se, but in a compacted schedule of events a reconnaissance well may quite likely become a monitoring well later in the program. This phase should therefore be considered part of the monitoring effort.

The variables to be included in the monitoring program depend both on the propagation mechanism, the technological feasibility to sample and the objectives of the monitoring program. The different objectives in the case of an accidental spill or leachate propagation from an unknown source are:

- Determination of the extent, nature and degree of contamination (source monitoring).
- Determination of the propagation mechanism and hydrologic parameters to initiate appropriate response.
- Detection and warning of movement into critical areas.
- Assessment of effectiveness of immediate abatement and cleanup operations.
- Recording of long-term evaluation for ambient trend monitoring and compliance with standards.
- Initiation of research monitoring to validate and verify models and assumptions on which response action was based.
- Initiation of case preparation monitoring for litigation or other regulatory activities.

Other objectives may be determined and added as the local situation warrants.

Contaminant Release Into the Environment

Anticipated and unexpected contamination

The two examples discussed in this article represent two extreme positions of a whole range of possible interactions of potential contaminants with the environment. Figure 2 illustrates the situation by the cross-marked fields. The diagram shows three types of potential sources for contaminants, namely direct disposal of wastes, the release of byproducts of manufacturing processes or land use-related operations (road salting for instance), and release potential of products during storage, transport or distribution. The contact relationship to the environment in general or to ground water flow systems in particular can be open, restricted or isolated. In the case of valuable raw materials (crude oils) or products, every incentive is given to limit wastage and spillage by isolation in pipelines or tanks. Release to the environment is only by accident.

The degree to which disposed wastes are isolated from the environment depends on their degree of hazard and magnitude of potential impact. Current municipal wastes are disposed in a relatively open environment. Certain types of industrial wastes are disposed into restricted environments such as subsurface injection sites. High-level radioactive wastes have to remain totally isolated from the hydrosphere and biosphere.

In planning, municipal waste disposal facility sites are selected on the basis of geologic and hydrologic studies. Monitoring systems are part of the planning and installation process. In case undesirable situations arise, early warning is given and countermeasures and contingency plans are readily activated. This of course is not the case in pre-regulation and illegal dumping sites and therefore special measures have to be taken. Basically these are quite similar to accidental spills along extended linear routes, where a high degree of detailed site planning and installation of monitoring systems is economically impractical. By contrast, localized isolation and storage facilities go through the same or an even more restrictive environmental planning and monitoring design process than solid waste disposal sites to anticipate and counteract accidental releases or spills.

Contaminant Propagation in Shallow Aquifer Systems

In line with the previous discussion about design and operation of a monitoring system, one must first determine the nature of the propagation mechanism and evaluate the physical properties of the release environment in order to plot the best strategy for field monitoring and related information-gathering processes.

Contaminant Propagation Mechanism

Two basic propagation mechanisms exist.

Water soluble contaminants: This is the case for all leachates generated from landfills which by definition are water soluble, although they may contain some oil and grease, stockpiles of soluble materials (road salts) or aqueous solutions (ammonia, liquid ferilizers, etc.).

A vertical cross section through a contamination case is given in Figure 3. Four separate zones are distinguished.

Zone I determines the surface configuration of the spill or the leached body. Liquid spills are governed by the local topography and migrate toward depressions that allow accumulation of the spill material. For leaching solids, the engineering and layout of the site and its relation to the surface drainage will determine where infiltration will take place. The areal extent and shape of Zone I determines the surface from which infiltration will occur and how much the contaminant will be spread.

Zone II is the soil zone, which may or may not be fully developed. It presents certain characteristics such as high retention capacity for the aqueous solution, adsorption and exchange capacity for specific ions, and the possibility of biological activity as a potential mitigating process.

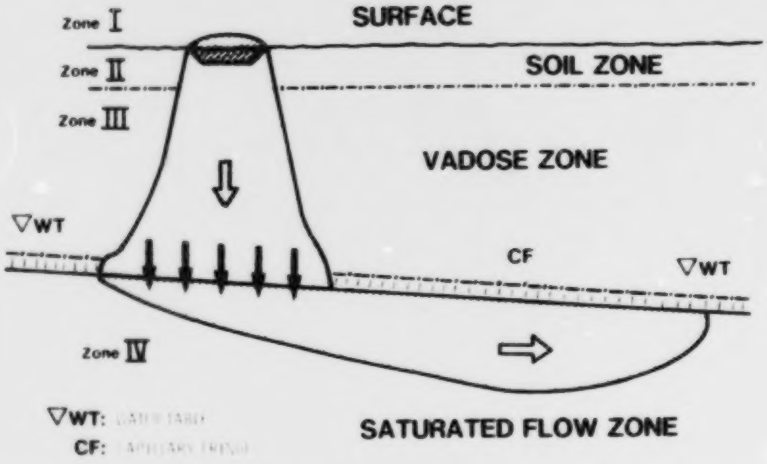


Figure 3 Subsurface propagation of a water-soluble contaminant

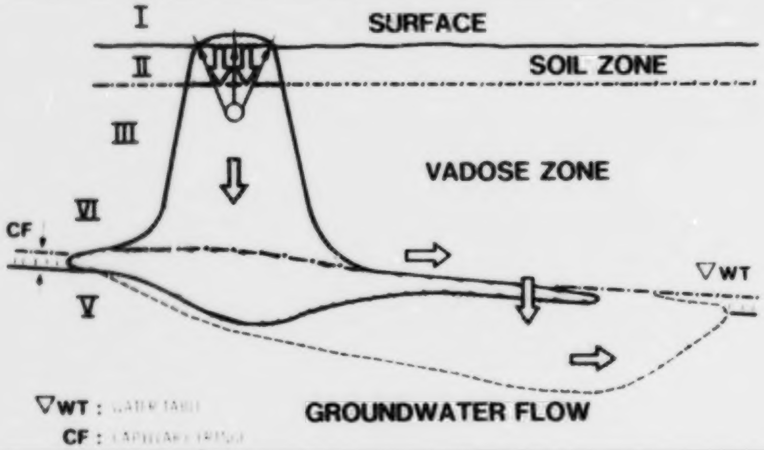


Figure 4 Subsurface propagation of a nonmiscible contaminant

Zone III represents the vadose or unsaturated zone where the solution is retained by capillarity to field capacity and where some adsorption of contaminants to the solid surface occurs. Depth to the water table or thickness of the vadose zone, areal extent of the spill as defined in Zone I and antecedent soil moisture and specific retention determine the volume of contaminant solution that can be held temporarily. If the spill volume exceeds this amount, the excess volume will enter the saturated zone and the ground-water flow field. In the case of soluble materials, each subsequent precipitation and infiltration event will displace the dissolved contaminant retained by capillarity movement downward. Changes in chemical environmental conditions such as pH or Eh may remobilize adsorbed contaminants. In a humid climate, all of the solute will eventually reach the ground-water body.

Zone IV is the ground-water flow field. As the dissolved contaminant leaves the vadose zone by downward vertical movement, it enters the capillary fringe and then the ground-water body. Under active flow it is transported laterally. The process at the interface between Zone III and IV is a simple entrainment of the constituents which, once having entered the ground-water flow system, are dispersed and diluted. A certain amount of adsorption on the grain surfaces can take place. This is the zone of the most active movement and potentially the one where the largest volume of ground water may be contaminated. It is therefore of utmost importance to determine ground-water flow directions and velocities and the dispersion characteristics of the aquifer materials.

Water insoluble or low solubility contaminants: Hydrocarbons and other organics belong in this category. These are the materials commonly released in pipeline breaks, tank car ruptures and reservoir tank corrosion. Although "oil and water do not mix," a small amount does dissolve into the water. In general, the amount dissolved is in direct proportion to the substance's volatility. Multiphase flow phenomena exist where immiscibility conditions prevail. Miscible or solution flow takes place where solubility condi-

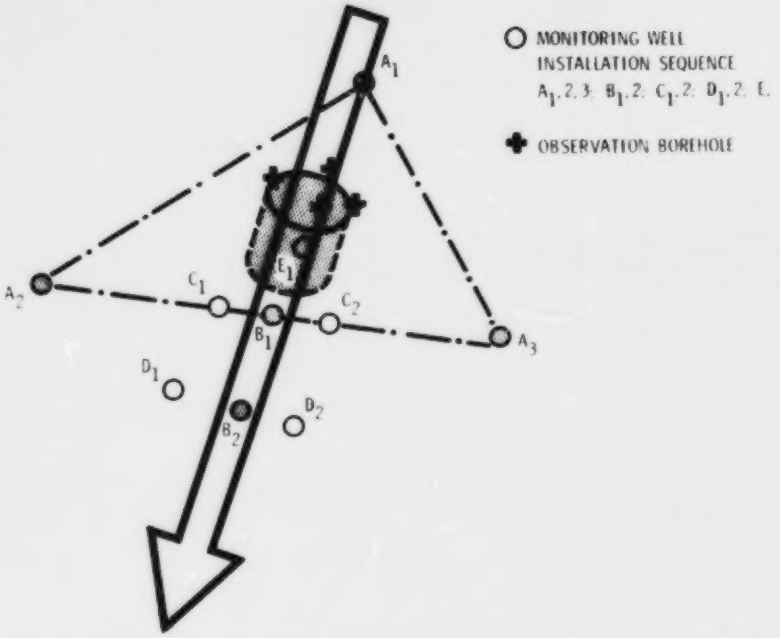

MONITORING WELL INSTALLATION SEQUENCE
A1, 2, 3; B1, 2; C1, 2; D1, 2; E.
OBSERVATION BOREHOLE
A1
A2
A3
B1
B2
C1
C2
D1
D2
E1

Figure 5 Drilling pattern around a spill site

tions prevail and both can exist at the same time. The mechanism of spreading is shown in Figure 4 where five different zones are distinguished, some of them directly overlapping with the different zones of the soluble case in the previous figure.

Zone I is the above-ground and surface zone where leaked or projected oil runs off and collects in surface depressions, thus forming the area from which infiltration takes place. This area depends on the local typography, the amount spilled and the conditions of release or eruption.

Zone II is the soil profile. From Zone I the oil starts infiltrating into the subsurface via the organic soil layer, if such a layer is present. This zone is characterized by its high organic content and high moisture content due to soil structure. If the soil is oleophilic, it has a much higher oil retention capacity than the underlying nonorganic deposits.

Zone III is the vadose or unsaturated zone. This is the most important zone for oil retention. Water saturates the pore space only partially and ranges in value from zero to field capacity. Oil, as the nonwetting phase, moves downward under the forces of gravity. At first it moves as a more or less continuous phase or "oil body," displacing excess water from the larger pores. When all oil has infiltrated from the surface the "oil body" will move downward by translation, but small amounts of oil will be left behind the trailing end, trapped as insular disconnected droplets. The oil body continues to move in a disintegrated fashion until all of the oil is trapped in the pore spaces of the vadose zone, if its total retention capacity exceeds the infiltrated spill volume. Any oil in excess of this total retention capacity reaches the ground-water body and spreads on the water table through the capillary fringe.

Zone IV is the capillary fringe which is partially water-saturated, directly connected with the ground-water body vertically, but discontinuous laterally. When excess oil reaches this zone it will spread laterally under its own hydrostatic pressure and form a lens on the water table. The spreading will come to a halt when the hydrostatic forces in the oil phase are counter-balanced by the capillary forces at the outer edges of the spreading oil lens. This movement is governed by the phenomena of relative permeabilities and multi-phase flow in porous media.

If the porous medium is homogeneous and isotropic and the water table horizontal, then the oil lens would be perfectly circular around the center of infiltration. In most realistic cases the water table has a slope which gives rise to an elliptically elongated lens extending in the direction of flow. The shape of this lens depends on the water-table gradient, ground-water flow velocities, the capillary properties of the multi-phase flow system, and the shape and orientation of the original infiltration area.

Zone V is the ground-water body. Most hydrocarbon compounds in a spill are lighter than water and therefore tend to float on the water table. Under the hydrostatic head of the continuous oil column an actual depression and penetration of the ground-water body below the water table occurs. This inverted mound will dissipate as the overlying oil body spreads laterally. The penetration and subsequent retraction may result in leaving trapped insular oil behind in the ground-water body. The most important feature of Zone V in the emplacement stage is the formation of an interface between the bottom of the oil lens and the free flowing ground water. It is at this interface that small but significant amounts of hydrocarbon compounds go into solution with the water and are spread by convective and dispersive transport mechanisms.

The two zones of greatest importance in spill propagation are the retention zone in the vadose zone and the dispersion zone in the ground-water flow zone. The retention zone determines how much of the contaminant will reach the shallow ground-water body and how fast it will move, hence determining how much time is available for corrective action. The dispersion zone determines the largest area the contaminant plume will occupy, the direction and intensity of migration and the ultimate persistence of pollution.

Physical Properties of the Release Environment

The hydrologic parameters necessary to evaluate the subsurface spreading of the contaminant are retention capacity of the materials in the vadose zone, hydraulic conductivity and the dispersivity and adsorption capacity of the ground-water flow zone. In the present discussion and also for the examples cited below, it is assumed that the aquifer materials are unconsolidated and granular, such as glacial or alluvial deposits.

Indirect information about these parameters can most readily be extracted from analysis and interpretation of grain size distributions of the porous materials. A preliminary assessment should be carried out in the field—more precise relationships have to be established in the laboratory.

Field assessment: The two types of samples available are grab samples (from the surface or from trench-ing operations) and borehole samples (when drilling is carried out). The grab samples will mainly give information as to infiltration capacity and characteristics of the vadose zone, whereas the borehole samples also give information about the hydrologic characteristics at and near the water table and in the saturated zone. The lithology and representative (average) grain size, sorting and homogeneity of the near surface materials is found from these samples.

Further field inspection of the topographic surface, the presence of standing water, and the extent of wetlands or open water bodies allows first approximate statements about infiltration capacities and subsurface permeabilities. This type of information serves as the basis to guide a more intensive sampling program.

If the spill involves the hydrocarbon products, their presence can be detected either visually or by smell. A preliminary idea about the extent of the subsurface infiltration can be gained by a physical inspection of the area.

Laboratory measurements: Measurements of grain size and distribution allow a more precise, albeit indirect estimate, of the characteristics of the porous media. Grain size analyses can be carried out rapidly to estimate permeabilities and hydrologic conductivities via the Krumbein and Monk (1943) equation. They also can be used to estimate retention capacity. For oil retention into originally dry soils, Convery (1979) has established an empirical equation in close analogy to the Krumbein and Monk procedure:

$$PR \quad 6.33\,(MGmm)^{0.75}\,e^{0.813\,\sigma g}$$

where PR primary oil retention
MGmm mean grain size in mm
σg graphic standard deviation

This equation is valid for a range of mean grain sizes between 0.5 mm and 0.177 mm.

A more site-specific but tedious procedure for obtaining specific or primary retention of oil or water is through drainage experiments in columns packed with representative materials from the site or borehole samples. The disadvantage of this method is that it is very time consuming, especially for very fine grain sizes, to the point where it becomes inapplicable as an information base for rapid decisions. Furthermore, properly designed experiments often need much larger volumes of materials than exploration borings will yield. Centrifuge methods have been developed (Eames, 1981) that allow an order of magnitude estimate of retention under various conditions of initial saturation with a small sample and within only a day or so of lab work.

Dispersivity measurements on packed columns in the laboratory have relatively little information value since, in general, field coefficients of dispersivity are one or two orders of magnitude greater than laboratory coefficients. This is due to the preponderance of inhomogeneities and preferential flow paths in the field situation. Laboratory experiments are useful only when adsorption rates are to be determined because the contaminant sees the same surfaces in the lab and the field. Experiments are time consuming. Their value is to corroborate or explain transport phenomena in the later phases of spill propagation. It is more important to get field data by installing an appropriate network of monitoring wells. These are also necessary to determine direction and velocity of ground-water flow.

Strategy of Field Monitoring and Information Gathering Procedures

Operations dealing with field monitoring procedures merge and become inseparable from the information gathering phase for systems behavior. The strategy now is to maximize the information content for both. The objectives for the combined phase are to assess the physical situation and the dynamics of the system and to obtain this information with the least but still adequate number of observation points or wells, all of which potentially could serve as monitoring wells in the later, more single purpose monitoring phase.

Preliminary information base: The general steps are:

- Rapid preliminary survey—Its objectives are to obtain an overview of surface drainage and ground-water drainage conditions, and of the extent and shape of the surficial contaminant source and its relation to the drainage system. Information for this survey is most easily obtained from topographic maps. Ground-water drainage patterns can be estimated from the slope and position of the topographic surface. Estimates are based on the hypothesis that the water table is a subdued replica of the topographic surface and that the position and elevation of water courses or water bodies are ground-water outcrops. Water-level surveys of nearby wells may also give an initial idea of flow directions. Grab samples of soil materials can be taken to assess the infiltration characteristics.

At the same time, a number of uncontaminated surface-water samples and supply-well samples far enough away from the spill site should be taken and analyzed for the principal contaminant. This is to establish a base of neutral background data. This preliminary information serves to design the information/monitoring well network layout.

- Intermediate and permanent observation points: The drilling of observation boreholes has as its main objectives, first to obtain information through borehole sampling, and second to actually install monitoring wells.

In the first phase, three wells have to be drilled in a triangular pattern to determine ground-water flow direction and hydraulic gradients near the contaminant site by the well triangulation method. One of the wells should be located far enough upstream of the site to serve as an uncontaminated control point. The other two will be downstream as indicated on Figure 5. Since accurate levels have to be established and since at least one of the wells will remain a permanent control, normal screen and casing installations, including proper surface seals, are needed.

The next phase consists of locating wells directly downstream on the flow line passing underneath the site of heaviest surface contamination or where greatest infiltration is assumed. If there are several areas of concentration, more than one such flow line has to be designated. On each flow line at least two observation wells should be drilled, one as close as possible to the extending plume, the other one at some distance downstream in uncontaminated materials. Thus the actual flow velocity of the contaminant and its propagation in the field can be found by measuring travel times of contaminant between the two observation points. In addition at least two wells if not two sets of wells should be constructed at right angles to the center flowline. This is to obtain an idea about the lateral spreading of the contaminant in the ground-water flow field. These observation points must be located as close to the advancing plume as possible without being inside of it; otherwise, unambiguous travel times cannot be established.

The extent of direct subsurface contamination can be determined by a number of boreholes serving only as temporary observation points. They are especially useful when the contaminant is easily recognizable in the soil sample from the borehole (split spoon or bailed sample), as for example hydrocarbons and other pungent inorganics. If recognition on sight is not that simple, other rapid field methods to analyze for the contaminant or an associated tracer become necessary. Depending on their position in the flow field and with respect to the plume, a decision can be made whether to case the hole as a permanent observation well or not. At least one well should be located in the area of high concentration in the center of the plume near the infiltration site but only extend to the deepest position of the contaminant. Its objective is to monitor the evolution of contaminant concentration in the spill area with time.

The first-phase wells which were drilled outside the area of direct contamination give information about till or clay layers or other barriers in the vicinity of the spill. It is of utmost importance to avoid drilling through these seals in subsequent drilling phases so as not to provide easy pathways of cross communication for the contaminant to enter either deeper aquifers or portions of the same aquifer that were naturally isolated from the infiltration site.

The third phase of drilling deals with obtaining better resolution on the basis of information gathered in the first two phases and with locating monitoring wells in critical or vulnerable locations.

For the wells constructed in the second and third phases, the necessary information should be available to decide at what depth the intake should be or whether or not multilevel sampling would be desirable. This is of special interest if the contaminant solution is of higher density than the ground water.

Toward the end of the third phase, it may be necessary to install a number of warning wells upstream of critical areas such as domestic or municipal supply wells. Although these wells may not necessarily be in a location that would be of the highest monitoring efficiency for the overall system, they serve as additional safeguards.

The soil and aquifer materials that need to be studied for their hydrologic characteristics are borehole samples from the vadose zone near the site of infiltration and samples from the yet-uncontaminated areas. The first samples are used to determine retention characteristics on the basis of grain size analysis and centrifuge experiments. The second samples need only be taken in the saturated zone for porosity and permeability determinations by grain-size analysis or by permeameter experiment. Both measures are important to establish a material balance for a spill or infiltration condition. Furthermore, the permeability-porosity data can establish independent estimates of ground-water flow velocities.

The soil samples need to be analyzed for traces of the infiltration material in order to establish the volume of aquifer material contaminated. The dissolved-phase water samples have to be withdrawn from the wells. This, however, deals with establishing the extent of the impregnation body and the plume at the time of emplacement. These samples should be taken as soon as possible after the contamination incident has occurred.

- The true monitoring phase begins after the basic information about the case and the hydrogeologic parameters have been at least partially completed. Its objectives are to follow the evolution and fate of the contaminant in the ground-water body, to serve as an early warning system in case sensitive areas are approached by the plume, to assess the effectiveness of cleanup and abatement operations, and to determine the final or closeout phase, after which no further adverse effects are expected from the incident and monitoring operations can be terminated. For certain hazardous and toxic waste sites, monitoring operations will have to continue indefinitely and no termination phase would be contemplated in that case.

Monitoring phase: The principal monitoring operation is subdivided into three phases.

1. The first phase is concurrent with the general information gathering process and is characterized by high-density and high-frequency sampling. All possible wells are used for sampling, all pertinent parameters will be measured and samples will be analyzed for a large number of constituents. Many environmental parameters such as hydrometeorologic and climatologic data will be gathered. The main objective is to rapidly recognize trends and general directions in order to more effectively design the next and more significant phase.

2. The second phase is the principal phase, specifically covering all of the objectives defined for the monitoring process. At this level, the results of the primary phase are evaluated, and a model concept of the system dynamics is formulated. A combination of the theoretical concepts and empirical data permits an assessment of the network layout and operational procedures. Application of statistics permits the establishment of confidence limits, minimum number of observation points in the system and number of samples to be taken.

The spatial level of resolution or number of monitoring wells depends on the preponderant mechanism of contaminant propagation. From a protection point of view, it also depends on the magnitude of the expected impact and the likelihood of its occurrence.

The first level of observation and analysis simply follows the evolution of the spreading pollution plume. This serves to verify whether the system actually behaves the way it has been assumed. In a more formalized and analytical context, results from this monitoring phase can be used to validate, verify and calibrate formal computer simulation models of contaminant transport and to adjust these models to the real conditions by curve-matching procedures.

Based on this information, abatement and cleanup procedures can be chosen among different alternatives. The monitoring process serves to indicate whether the remedial actions are in the right direction, how effective they are, and how close the system performance comes to meeting predetermined or statutory standards.

3. Long-term and final monitoring phase. With respect to time horizons, the monitoring system operator may be interested in daily, weekly or monthly averages, or he may follow events on a seasonal basis or on an event-by-event basis. The latter deals with total reactance of the system to a precipitation and infiltration event which manifests itself in water-level fluctuations in the well hydrograph and in concentration fluctuation due to dilution, displacement or remobilization of retained or adsorbed phase contaminants.

At this point, decisions are made regarding which monitoring wells should operate in the long-term or final phase of monitoring and at what levels the entire monitoring and surveillance operations can be terminated. It is conceivable that this process may be staggered over a relatively long period of time with some key wells operating on a quarterly or semi-annual basis and finally only one or two wells being checked on a yearly or longer interval. This phase should also look into the last leg of the rehabilitation and restoration phase, namely the removal from service of the monitoring network when its purpose has been served.

Materials and procedures: In all three of these phases, proper attention to selection of materials, installation and operation of the monitoring network has to be given.

The materials such as well casings, screens, valves, conduits, etc. have to be compatible with the nature of the contaminant. They should not adsorb to or react with the surfaces they encounter, nor should the surfaces release compounds that would interfere with the analysis or directly change the concentrations measured.

The sampling method also should take into account the nature of the contaminant and the conditions at the sampling point. Vacuum methods and contact with the free atmosphere should be avoided when the material is highly volatile. Gas lift pumps should operate with inert gases if any reaction with the contaminant or with the solute is expected. Environmental conditions representative of the sampling point environment should be maintained to give the same solubility conditions lest precipitation, concentration or evaporation occur. In the case of two-phase conditions such as a floating layer of hydrocarbons on the water surface in a well, care has to be taken to either measure it separately or to obtain an undisturbed sample using a bottom sampling device. Simple pumping in this case only draws in supernatant oil from around the well and the sample may over-represent the actual free oil phase and the dissolved phase which may have been increased by turbulent mixing in the well bore.

All materials, instruments and sampling procedures should be geared to analytical methods used for the final analysis and the special volume, handling and transportation conditions that guarantee the greatest efficiency.

Field Examples and Applications

To amplify the hitherto more general discussion, two practical cases are presented. The first deals with a pipeline spill and falls into the category of an accidental release of a product from an isolated environment (see Figure 2). It basically has the same random characteristics as accidents from ruptured railroad tank cars or tank trucks. The difference is that in batch transportation, spilled volumes are much smaller than those from a pipeline break. On the other hand, overland pipelines never carry highly toxic or extremely hazardous materials, as often is the case with tank trucks.

The second case deals with contamination from stationary sources where ignorance about existing conditions at the time of contaminant detection introduce an element of chance, to treat them as quasi-accidents.

Monitoring a Pipeline Spill

The following describes the monitoring and information gathering processes after a pipeline spill in north-central Minnesota. About 10,000 bbl of crude oil had been released by flow, eruption and spray into the environment. The site is typical for the glaciated terrain of central Minnesota. Specifically, the spill is located near the margin of a small outwash plain with moraine deposits.

Immediate response and site inventory: The first steps in managing the spill were to eliminate any hazardous conditions, stop the flow and contain the spill. Thereafter a detailed small-scale topographic map of the area was established by professional surveyors. On this map, the areal extent of heavy oil concentration that had accumulated in depressions, intermediate concentrations along flow paths, and light concentration in the sprayed or dusted portions was indicated. The map served further to exactly locate future boring and well positions with respect to the pipeline and established benchmarks.

Immediately, eight observation borings were drilled to outline the areas of heaviest oil infiltration. Within these, shallow collection trenches were dug and the inseeping crude was recovered by pumping.

The general position of the site with respect to lakes, ponds and surface drainage courses as well as the slope of the land surface gave preliminary indications about expected ground-water flow directions. At first, two observation well triangles were installed around the two most contaminated sites, one right at the rupture, the other near a depression into which a large amount of oil had drained and collected.

Water samples were taken immediately after installation for analysis, water levels were measured, and the altitudes of the top of the well casings were surveyed. On the basis of this information, tentative flow directions could be established. It was decided to fill in the network with more wells in order to draw equipotential lines of water table elevation and to construct a flow net for the spill area. This was done because of the large extent of the affected area.

Also, in the very beginning, 35 samples of surface water were taken around the area but sufficiently far away so that no contamination by spray from the spill could be assumed. This was done to obtain a suite of background water quality samples. The area has peat bogs and other possible sources of organic materials that might give high readings of organics. Of the 35 samples taken, each showed that those organics of concern were below detection limits.

Twenty-six boreholes were drilled, and 15 were installed as monitoring wells. Drill site location and the decision to use the boreholes as monitoring wells were made in incremental steps, based on the finding of the previous two or three borings.

During the design phase for well installation, recovery by pumping proceeded and about 6,500 bbls of oil were ultimately recovered. Visual inspection of the collection trenches seemed to indicate that the oil seepage was mostly controlled by inhomogeneities of the soil materials, which offered preferred and restricted paths of migration, and also provided the opportunity for some lateral movements in the vadose zone.

Boring samples ranged from totally oil-saturated to only a few percent saturation. In many cases, a low concentration layer might be located between an upper and lower highly saturated layer, confirming the role of heterogeneities in oil migration. Grain size analysis by sieving showed the generally sandy materials to range between medium-coarse to fine sands with hydraulic conductivities ranging from $K = 10^{-4}$ to $K = 10^{-2}$ m/sec.

Hydraulic gradients were established from the flow net. Gradients were typical for this type of terrain, about 17 ft. per mile or 3.33×10^{-3}. Darcy fluxes of q between 3.3×10^{-7} and 3.3×10^{-5} m/sec, and actual velocities assuming a porosity of about 0.3 of $v = 1.1 \times 10^{-7}$ to 1.1×10^{-5} m/sec were calculated.

These figures are of importance in determining spreading and dispersion since the degree of mixing depends on the actual flow velocity in the pores.

As plastic-based pipe and casing materials may interact with and/or adsorb organic contaminants, only steel pipe was used for well casing. Some hydrocarbons from the coating on the steel casing must have bled into the well water and contaminated early samples, even though they were well cleaned. Subsequently, only galvanized pipe was used, and all of the existing wells were steam-cleaned in the field.

Sampling and monitoring phase: In most wells, the water table was below the vacuum-lift range. No sophisticated sampling devices for small-diameter wells were on the market at the time, so a somewhat simplistic bailing method was used. If a floating layer or sheen of free-oil phase exists on the water surface in the well, repeated dipping of the bailer leads one to overestimate the amount of free oil in the well. Also, by mechanically mixing this surface layer into the bulk of the sample, higher dissolved oil concentrations than are actually present in the undisturbed aquifer may appear. Concentrations are reported as measured, but because of the above-mentioned situation, they may be higher than actual concentrations.

The State Pollution Control Agency required gravimetric determination of freon-soluble oil and grease by separatory funnel extraction (Storet No. 00556). The wells were sampled frequently in the beginning and with longer intervals subsequently. Most of the samples showed values around the lower end of the detection range of this method, approximately 5 mg/l. The precision of the method is also lowest here. Plus or minus 1 mg/l in this range represents a relative error of about 20 percent. Through statistical analysis and manipulation, the information content of the results was improved. The results of the statistical evaluation were used to identify wells with problem concentrations, to separate the dynamic effects from random effects, and to select the most critical wells for the extended monitoring phase. The first criterion for selection is the average concentration for all samples in a given well. Any values of 5 to 6 ppm or below were considered noncritical. Values between 6 and 10 ppm indicate an intermediate concentration determined with low but adequate precision, values of 20 ppm and above clearly indicate contamination. In order to rank the wells in a semi-quantitative way, a "problem index" (Ip) was developed. It consists of the product of the mean, the variability, and the maximum ratio. The latter is the highest value in a series divided by the mean to normalize it. Values clustered around indices of 1-2 and up to 10 (control), around 10 to 15 (borderline), and above 50 (critical). This became the base of selecting the ultimate sampling network of control, borderline and critical wells. In some cases, Wilcoxon W-statistics (a ranking procedure) were used to clarify borderline cases using control-well data and data from suspected contaminated wells.

Oil retention studies of the average soil material from the site were concurrently carried out to calculate the total retention capacity of the vadose zone beneath the site. For the final analysis, centrifuge data of specific oil retention were used for balance calculations. Penetration of free oil phase to the water table was possible only through preferred channels or under extremely heavy surface concentrations which meant that only a minimal amount of crude could have reached the ground-water system. No free oil on the water table was detected in observation wells except directly beneath the most concentrated spill sites.

Abatement procedures parallel the monitoring phase. Free oil spray was burnt off the vegetative surface; 10 percent to 20 percent of the oil had evaporated and about 10,000 cubic yards of oil-contaminated soil were excavated from the sites that monitoring operations had indicated were especially critical. The excavated material was spread over 15 acres, fertilized and cultivated to permit rapid bacterial decomposition.

Oil and grease readings have been declining steadily. The number of wells monitored has been reduced to half, and frequencies now are at a quarterly interval and should soon be terminated.

Pre-regulation Land Disposal Sites

Site inventories: The state of Minnesota, especially the St. Paul-Minneapolis metropolitan area, may have potential problems with contaminant release from pre-regulation or unregulated (illegal) waste disposal sites. The problem becomes especially obvious when urban development reaches into areas with unanticipated problems.

Sites giving rise to problems range from those whose location and contained material are known, but for which no specific hydrologic data exists to clandestine or illegal disposal operations of actually hazardous wastes. In between lie a whole range of other combinations of known locations but unknown materials, unknown or forgotten sites, approximate locations with some vague ideas about the input, etc. Many fill operations have also operated as burning dumps so that the character of the input has drastically changed.

Common to all of these situations is the lack of information about their hydrogeologic and environmental impact. When a problem surfaces, usually by contamination of a drinking water supply, then all of the same methodology used in accidental spill monitoring has to be deployed. This has the disadvantage that little time is available for planning and making rational decisions. The main objectives are to protect the water supply, to identify the source and type of contaminant, and then to plan development of alternative supply sources or the abatement and removal of the contamination source. Operations are usually costly, can often not be assessed to the responsible party, and stir public opinion.

In order to address the problem, the Minnesota Pollution Control Agency (MPCA) has instigated an open dump site inventory in compliance with section 4005(b) of the Resource Conservation and Recovery Act. It identified about 1,400 sites outside and 120 in the metropolitan area. In addition, the MPCA has established a task force that reacts immediately to reporting of contaminants and also acts on statewide information about past or present improper disposal practices. The objective of this inventory is to direct information gathering operations at the critical sites so that problems may be anticipated. The most important problem at this point is to rank the known sites and to identify those that need immediate attention. Thus, resources and effort for hydrologic investigations and monitoring installations would go to those sites that most probably have or will develop critical pollution problems.

Development of ranking systems: The hydrogeology group at the University of Minnesota was involved in establishing such criteria for metropolitan sites.

The MPCA has a ranking system based on federal guidelines from top to low priority including those that are in compliance with statutes and that probably will not emit contamination. The technical ranking is according to degree of pollution potential. The system is not explicit enough from a technical point of view to avoid ambiguities and subjectiveness of surveyors.

The more extensive theoretical criteria that were evolved deal with:

1. Infiltration and dispersion characteristics of the hydrogeologic environment.
   - Surface drainage pattern around site.
   - Infiltration characteristics of soil.
   - Retention capacity of unsaturated zone.
   - Depth to water table.
   - Ground-water flow pattern and activity, including the existence of preferred flow patterns such as in fractured or cavernous rocks.
   - Interaction pattern of surface water bodies and ground water.
2. Impact characteristics.
   - Waste characteristics
     - Nature of wastes (toxic, hazardous, inert)
     - Volume and area covered
     - Solubility characteristics
     - Residence time since emplacement
   - Human geography characteristics.
     - Degree and nature of development (density, i.e., urbanized residential, industrial, farm, etc.).
     - Land ownership as a measure of ease to initiate response activities (public, state, municipal, private, etc.)
     - Relation of flow path to ground water use (intensity of withdrawal and customers served).
     - Relation of flow path to environmentally sensitive areas.

In all of these criteria, it is easy to qualitatively indicate relative severities: the deeper to the water table the better, the higher the retention capacity the better, the lower the toxicity and the lower the density of affected users the better, and so on. However, it is very difficult to assign uniformly applicable numerical values to these situations. Even before it reaches this point, other difficulties due to insufficient information show up.

The first step in assessing pollution potential is to obtain information about the hydrogeology of the site. Search of files, existing well logs, and records in city and county engineers' offices are necessary to get some background information. Site inspection and reference to topographic maps offer some clues about potential infiltration, ground-water flow directions, and the relation of the site to surface drainage systems. This is about as far as a preliminary survey can go. At most, three categories can be established:

- Visual evidence of pollution by leachate.
- Site within or very near the saturated zone.
- Site normally above water table.

Actual field work is needed for more detail especially since situation one, the intuitively more offensive one, might have less impact than an undetected severe condition under situation three.

Determining anything about waste volumes and their degree of toxicity or hazard is almost impossible for pre-regulation landfills, where no logs were required. Some information can be obtained by interviews of residents or former operators about possible sources for the wastes and about the management practices during active operations.

Another approach is to reconstruct the industrial or commercial waste-generating potential within the area served while the facility was operating. The assumption is that if no hazardous wastes were generated in the area it is unlikely that any large amounts are to be found in the dump site. Information about past industrial activities can be obtained through chambers of commerce and state agencies dealing with economic development and activities.

Human geography data are the easiest to come by. Information such as location of high density residential areas, municipal ground-water withdrawal sites and volumes, types of land use and degree of development is relatively easy to obtain. Much information can be gathered at the municipal or county level and, in areas with effective regional government such as the Twin Cities Metropolitan Council, much of this information may already have been collated and put into directly usable form.

Sample assessments of 25 sites in the metropolitan area resulted in a simplified and reduced list of workable ranking criteria. The first group deals with the actual knowledge of suspicion of hazardous wastes in the site. This is amplified if management control and general maintenance have been lax. The next two groups are almost equal in importance for ranking, but they are weighted according to the amount of information available for each and its reliability. The two groups are: a) the hydrologic conditions as measured by the three categories: position near water table-above water table indicated above, and b) the bulk of the demographic and user information that constitutes the severity of any potential impact from unexpected contaminant releases. It is on the basis of this information that resources for more detailed study, field investigation and the development of monitoring networks around abandoned waste disposal sites should be allocated. Once the site is studied in depth, some of the critical aspects may shift and, for example, it might be shown that the hydrogeologic situation creates a serious problem although the original input into the site was less hazardous than in other comparable sites.

Summary

Accidental releases and unexpected detection of contaminants requires an immediate response to mitigate and eliminate the resulting pollution. Management decisions and subsequent responses have to be carried out rapidly. The information phase to study and present alternatives for decision making and for the monitoring process run concurrently rather than sequentially as in planned and controlled containment and management schemes. Therefore, it is necessary to obtain information rapidly and to maximize its efficiency by conjunctive use between the different sectors of the management/decision process. For the monitoring phase, this is done by incremental collection of information, immediate evaluation and dynamic incorporation in the network design process.

Acknowledgement

This study was partially supported by the Department of Interior, under project A-041 through the Water Resources Research Center, University of Minnesota, as authorized by the Water Resources Act of 1964, public law 88-379.

References

Convery, M.P. 1979. The behavior and movement of petroleum products in unconsolidated surficial deposits. M.S. Thesis, University of Minnesota, Minneapolis.

Eames, V.A. 1981. Study of oil retention under field and laboratory conditions. M.S. Thesis, University of Minnesota, Minneapolis.

Krumbein, W.C. and C.D. Monk. 1943. Permeability as a function of the size parameters of unconsolidated sand. AIME Trans., v. 151, pp. 153-163.

Pfannkuch, H.O. 1975. Study of criteria and models establishing optimum level of hydrogeologic information for ground-water basin management. Bulletin 81, Water Resources Research Center, University of Minnesota, Minneapolis.

Pfannkuch, H.O. and B.A. Labno. 1976. Design and optimization of ground-water monitoring networks for pollution studies. Ground Water, v. 14, no. 6.

Todd, D.K., R.M. Tinlin, K.D. Schmidt and L.G. Everett. 1976. Monitoring ground-water quality-monitoring methodology. Report EPA-600/4-76-026, Las Vegas.

Biographical Sketch

Hans-Olaf Pfannkuch attended the University of West Virginia School of Mines from 1952-53, received his Diplom-Ingenieur in Mining from Rheinisch-Westfalische Technische (Germany) in 1957, and his Ingenieur-Docteur from the Universite de Paris in 1962. Since that time, he has served as research engineer at the Institut Francais du Petrole, as assistant professor at both the University of Illinois and Southern Illinois University, and as associate professor at the University of Minnesota, his current position. He has written several text books including Elsevier's Dictionary of Hydrogeology (1969), and journal articles on subjects ranging from petroleum well logging to design and optimization of ground-water monitoring networks. While at the University of Minnesota, he has been active in a wide variety of applied research projects dealing with ground water and surface water and with information requirements for managing and monitoring ground-water systems. As a consultant, he has been involved in field studies and network design problems of spills due to pipeline breaks in central and northern Minnesota.

MONITORING IN THE VADOSE ZONE

By L. G. Wilson

Introduction

The geological profile extending from the ground surface to the upper surface of a principal water-bearing formation is called the vadose zone. As pointed out by Bouwer (1978), the term "vadose zone" is preferable to the often-used term "unsaturated zone" because saturated regions are frequently present in some vadose zones. The term "zone of aeration" is also often used as a synonym for vadose zone. In some regions the vadose zone may be only a few feet in thickness. Elsewhere, the vadose zone may be tens or hundreds of feet thick. Changes in the thickness of the vadose zone at a given site reflect the response of a water table to either recharge or discharge. Figure 1 is a cross-section through the vadose zone of an alluvial basin in the west.

Monitoring within the vadose zone underlying a surface pollution source affords an "early warning" of potential ground-water pollution. If remedial measures are implemented prior to the onset of pollution, the associated renovation costs will be eliminated or reduced. Concomitantly, vadose zone monitoring may reduce the need for extensive ground-water monitoring. In other words, if a vadose zone monitoring program fails to detect the movement of pollutants, the requirements for ground-water monitoring may be reduced or largely precluded. The savings in costs for constructing ground-water monitoring wells could be significant, particularly in western regions where water tables are often hundreds of feet deep.

Vadose zone monitoring is appropriate for a number of potential pollution sources including sanitary landfills, land treatment facilities, spoil piles, septic tank areas, pits, ponds, lagoons, dry channels used for effluent disposal, and irrigated fields. The "Hazardous Waste and Consolidated Permit Regulations" issued by the Environmental Protection Agency (1980) require that areas used for land treatment of hazardous wastes shall install a vadose zone monitoring system, in addition to ground-water monitoring wells. However, vadose zone monitoring was not recommended for hazardous waste landfills and surface impoundments due to the problem of installing sampling units, such as suction cup samplers, beneath existing facilities. Because of the importance of obtaining an early warning of pollutant travel from impoundments and landfills, however, alternative possibilities should be considered. For example, sampling units could be installed during construction of new facilities. Similarly, sampling and nonsampling units could be installed on the perimeter of existing disposal facilities without disturbing the waste deposits. Sampling units are defined as those providing liquid or solid samples for analysis. Nonsampling units provide inferential evidence of liquid/pollutant movement.

Basically, a monitoring program for the vadose zone comprises a premonitoring phase followed by an active monitoring phase. Implementation of both phases will draw upon an array of sampling and nonsampling techniques for characterizing three properties: (1) storage (and storage changes), (2) fluid transmission and (3) pollutant mobility. Information on storage properties is important in that liquid wastes are placed in temporary storage within the void space of the vadose zone prior to movement into ground water. In some cases, such as in western valleys, the storage space may actually be sufficient to preclude movement into productive aquifers. In fact, Winograd (1974) suggested using the vadose zone for the storage of low-grade radioactive wastes. Mann (1976) recommended using the storage space of the vadose zone in arid regions to augment the above-ground storage capacity of tailings ponds. In addition to an interest in storage, *per se*, information on storage changes is of prime importance in a monitoring program. For instance, a change in the liquid content of the vadose zone at a given depth beneath a disposal facility suggests that liquid-borne wastes have reached that depth.

The transmission of liquid wastes in the vadose zone is expressed in terms of flux and rate of flow. "Flux" (actually, the term "darcian flux" is more accurate—see Freeze and Cherry, 1979) represents the quantity of liquid crossing a unit cross-sectional area per unit time. Information on the flux of liquid wastes can be used to estimate the amount of liquid-borne pollutants entering a ground-water system. Similarly, estimating the velocity (or "average linear velocity"—see Freeze and Cherry, 1979) of liquid wastes in the vadose zone is useful for establishing a lower limit on the arrival time of pollutants at the water table. This provides a conservative estimate of when pollution might begin. The actual travel time of specific pollutants may be longer because of attenuation. Unfortunately, very few field measurements have been obtained on either the flux or velocity of liquid wastes in representative vadose zones. Evans, Sammis and Warrick (1976), reported deep percolation rates for an irrigated field in the Salt River Valley, Arizona. Estimates of the darcian flux ranged from 9 to 38 cm/yr (4 to 15 in/yr). Corresponding estimates of the average linear velocity ranged from 57 to 130 cm/yr (22 to 51 in/yr).

Specific pollutants may not necessarily travel at the same rate as the fluid in which they are entrained. Actually, the vadose zone may attenuate certain pollutants to the point that they could be excluded as pollution sources. Runnells (1976) demonstrated that soil from Sulfur Spring, New Mexico, has an enormous capacity to remove copper from copper mill wastewater. The additional observation that copper removal was irreversible indicated that thousands of years would be required before ground water (approximately 100 feet deep) at the site would be affected. Thus the purpose of a sampling program in the vadose zone is to determine the mobility of particular pollutants, as moderated by attenuation mechanisms.

The present array of tools for monitoring in the vadose zone consists of existing techniques from related disciplines such as ground-water hydrology, soil science and geophysics. For example, approaches developed by soil specialists for characterizing the storage and flow of soil water and solutes in the root zone of crops have been extended into deeper reaches of the vadose zone. Similarly, techniques employed by hydrogeologists for monitoring ground water are used for monitoring perched ground-water regions of the vadose zone. Unless an interdisciplinary approach is undertaken, the task of designing a vadose zone monitoring program for a pollution source may be assigned to a specialist in a particular discipline. Such an individual may be unfamiliar with the broad array of alternative methods.

The purpose of the three articles in this series is to provide a primer on techniques for vadose zone monitoring. This article deals with methods for determining storage and storage changes; the second will discuss methods for characterizing the transmission of liquid wastes; and the third will cover techniques for characterizing pollutant mobility. In each article an elementary technical review will be presented, followed by a detailed examination of a popular method. Finally, a catalog of common, alternative methods will be included. This catalog summarizes the principles of operation, together with major advantages and disadvantages of each method. Appropriate literature citations are included for those readers seeking additional information on each method. The third article will demonstrate how the individual methods could be grouped in a premonitoring/monitoring program.

Unless specified, the carrier fluid associated with waste transmission is assumed to be water.

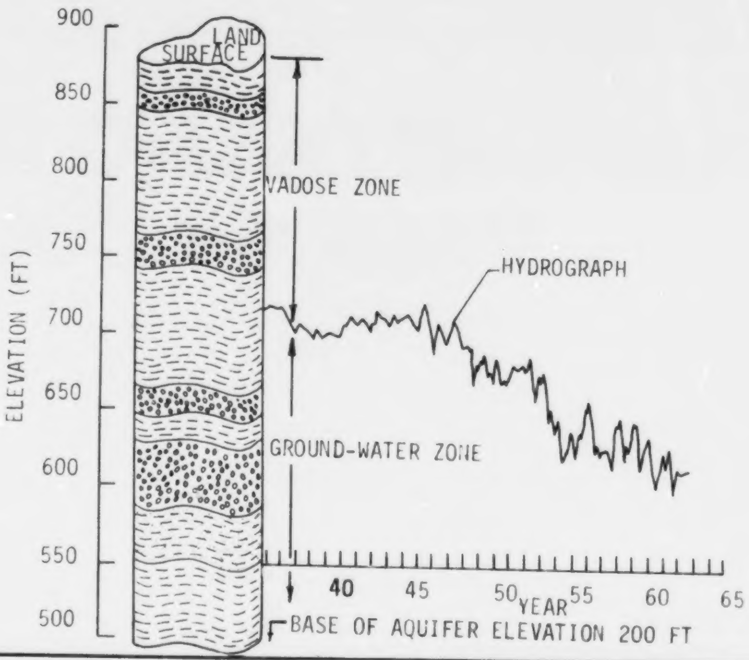


Figure 1. Cross-section through a vadose zone in an alluvial valley of the western U.S. Note that the thickness of the vadose zone changes in response to pumping or recharge (after Ayers and Branson, 1973.)

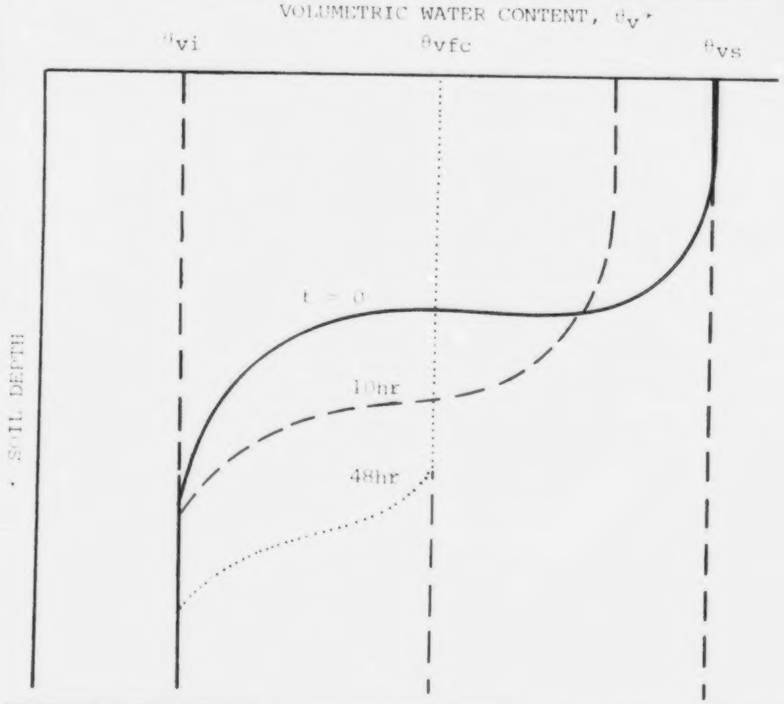


Figure 2. Idealized water content profiles in a soil for three different times. The curve for t = 0 represents the profile at the instant surface application of water is stopped. Redistribution of infiltrated water occurs as shown until the profile approaches field capacity (t = 48 hr) (from Hanks and Ashcroft, 1980)

Storage Changes

The total storage capacity of the vadose zone at a given site is a function of the total thickness of the zone and the porosity of the associated media. As shown on Figure 1, the thickness of the vadose zone, and consequently the overall storage capacity, can change in response to recharge or discharge conditions. In addition, the available storage capacity cannot be equated to an overall or representative porosity value because residual water generally occupies some of the storage space. Residual water may constitute water present in the pore space during deposition of vadose zone materials; water remaining following recession of the water table; or water arriving by deep percolation from the land surface. The effect of the presence of residual water is to diminish the total available storage capacity of the vadose zone for liquid wastes.

Infiltration of ponded water at the land surface increases the volume of water stored in the uppermost horizons of the vadose zone. A "snapshot" of a water content vs. depth profile in a homogeneous medium during infiltration, such as depicted on Figure 2, shows that water content values approach saturation for some distance in the profile but diminish rapidly to initial values near the wetting front. If infiltration is eliminated or greatly reduced (such as by clogging of surface pores) water becomes "redistributed" throughout the profile as water continues to move downward by gravity (see Figure 2). The amount of fluid stored above the gradually deepening wetting front approaches values that are greater than the residual value, and less than or equal to those corresponding to "field capacity."

By appropriate equipment (e.g., neutron moisture logger), the magnitude of the volumetric change in storage, $\Delta\theta$, can be determined. The same equipment can be used to estimate the depth, D, of the wetting front. Knowing these values, the depth, d, of fluid crossing the surface can be calculated by the following equation:

$$d = \frac{\Delta\theta}{100} D$$

For example, if the depth of penetration of a liquid waste is 100 feet, and the measured change in water content is five percent by volume, the depth of applied fluid would be 5 feet.

Other physical properties relating to the storage of liquid wastes in the vadose zone are: (1) the relationship between water pressure and water content, (2) field capacity and (3) fillable porosity. In an unsaturated medium, water is held by various forces within the pore space (or fractures) at water pressures less than atmospheric. Hanks and Ashcroft (1980) reviewed such forces and related potentials in detail. The potential related to the adsorptive forces in a soil matrix is called the "matric" potential (Hanks and Ashcroft, 1980). The relationship between water content and matric potential for a given material is depicted by a "water characteristic curve." Such curves are prepared in the laboratory from analyses of core samples or disturbed solids samples using a pressure plate apparatus (see Richards, 1965) and in the field using a neutron moisture logger (see Bouwer, 1978). Figure 3 includes water characteristic curves for various soils. Water characteristic curves show that as the material becomes unsaturated—for example through drainage—water content values steadily decrease and corresponding pressure values increase negatively. As seen on Figure 3, materials of different texture have different curves. For example, the curve for a clay is displaced to the right of that for a sand because water is held more tightly in finer textured material. Also, different curves will be obtained for the same material depending on whether it is being wetted or drained.

Field capacity (specific retention) may be defined in a general sense as the volume of water which a unit volume of soil will retain against the force of gravity during drainage. The concept of field capacity was developed many years ago by agriculturists concerned with quantifying the amount of water to apply to irrigated fields. The original premise was that field capacity is a fixed value representing the amount of water stored in a soil a certain time after drainage has "essentially ceased." By the same token, it is usually assumed that during recharge (wetting), water movement will not occur until the medium has been wetted to field capacity. Although these concepts of field capacity are useful in an applied sense, one should be aware of certain technical limitations (Hillel, 1971).

One limitation is that the simplistic concept of field capacity fails to account for the dynamic nature of soil water movement. In particular, drainage does not really cease at field capacity but may continue at a slower rate for a prolonged period of time (Hillel, 1971). The modern view

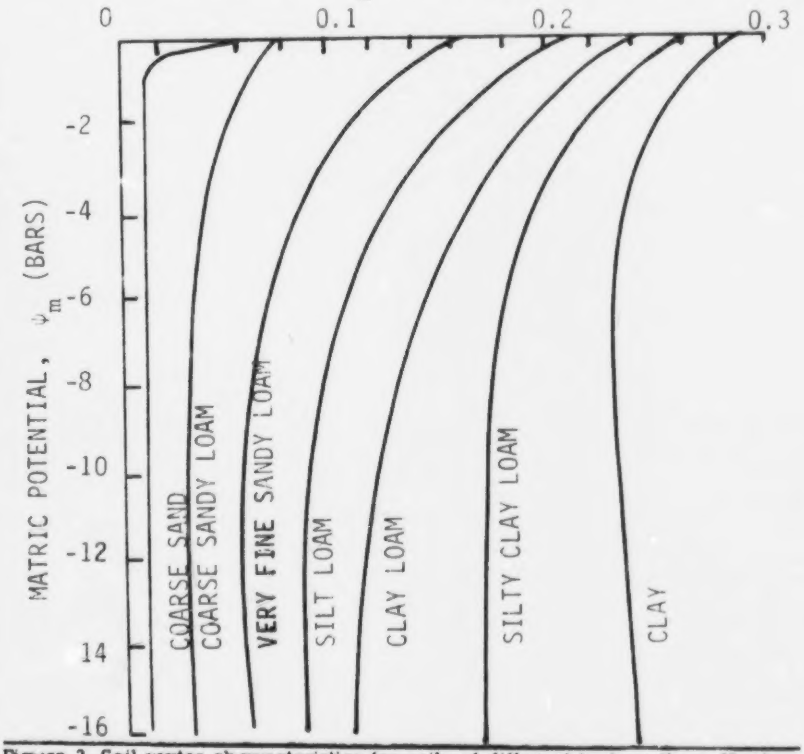


Figure 3. Soil water characteristics for soils of different texture (from Hanks and Ashcroft, 1980).

of field capacity is that it is not a unique soil property; instead, a range of values is possible.

The volume of water that an unconfined aquifer stores during a unit rise in water table per unit surface area is called the fillable porosity (Bouwer, 1978). The amount of water placed into storage (fillable porosity) during the rise of a water table is less than the corresponding amount released during drainage (specific yield). The difference in storage reflects hysteresis caused by air entrapped in pore sequences during the rise of the water table.

Methods for Monitoring Water Storage and Storage Changes in the Vadose Zone

Determining the overall thickness of the vadose zone at a site requires measuring the depth to ground water (for water table conditions) or determining the depth to the base of a confining layer for artesian conditions. Techniques for measuring depths to water include electric sounders, tapes, bubbler techniques and neutron moisture logging. Changes in water levels are monitored using automatic water stage recorders. Drillers logs or geological logs may also be useful in determining the depth to confined ground water.

Methods for monitoring water storage and storage changes in the vadose zone include a direct method, designated the gravimetric method, and indirect methods such as neutron moisture logging, gamma ray transmission, gamma ray scattering, tensiometers, electrical resistance blocks, and thermocouple psychrometers. The principles, advantages and disadvantages of each of these methods, as well as other methods such as remote sensing, were extensively reviewed by Schmugge, Jackson and McKim (1980). The catalog of methods, presented in the latter part of this paper, also summarizes alternative methods. Only the neutron moisture logging method will be discussed in detail in this section.

Neutron Moisture Logging

Agriculturists have employed the principle of neutron moderation or thermalization for many years to measure the volumetric water content of soils to assist in scheduling irrigations. The technique has also been used in water resources investigations to estimate the specific yield of unconfined aquifers; to locate ground-water tables; to determine the porosity of water-bearing formations; and to develop water content profiles in the vadose zone (Keys and MacCary, 1971). Keys (1967) used the neutron logging method in conjunction with natural gamma logging to delineate perched ground water above a clay bed at a site on the National Reactor Testing Station in Idaho. Wilson and DeCook (1968) examined neutron logs in a transect of access wells perpendicular to the Santa Cruz River, Arizona, to estimate lateral flow rates in a perching layer during recharge from ephemeral flow events in the river.

The neutron thermalization method is based on the principle that high energy neutrons are slowed down (thermalized) in their trajectories primarily by collisions with light atomic nuclei, especially those in hydrogen atoms. Hydrogen atoms are mainly present in water molecules within the vadose zone pore space. The density and radius of the cloud of thermalized neutrons will thus be a function of the water content of the media and the strength of the fast neutron source (Brakensiek, Osborn and Rawls, 1979). The principle of operation of a neutron moisture logger is depicted in Figure 4. Readers interested in technical details on the principles of the method should consult references such as Schmugge, Jackson and McKim (1980), Brakensiek, Osborn and Rawls (1979), Gardner (1965), and van Bavel (1963).

Instrumentation used to measure water content by neutron thermalization requires three principal components: (1) a source of fast neutrons, (2) a detector of slow neutrons and (3) an instrument to determine the count rate from the detection equipment. These components are mounted within a cylindrical probe, such as that shown on Figure 4, which can be lowered into a cased or uncased borehole or an access tube. A common source of fast neutrons is americium-beryllium. Smaller units used by agriculturists contain sources in the millicurie range (e.g., 50, 100, 150 millicuries), requiring

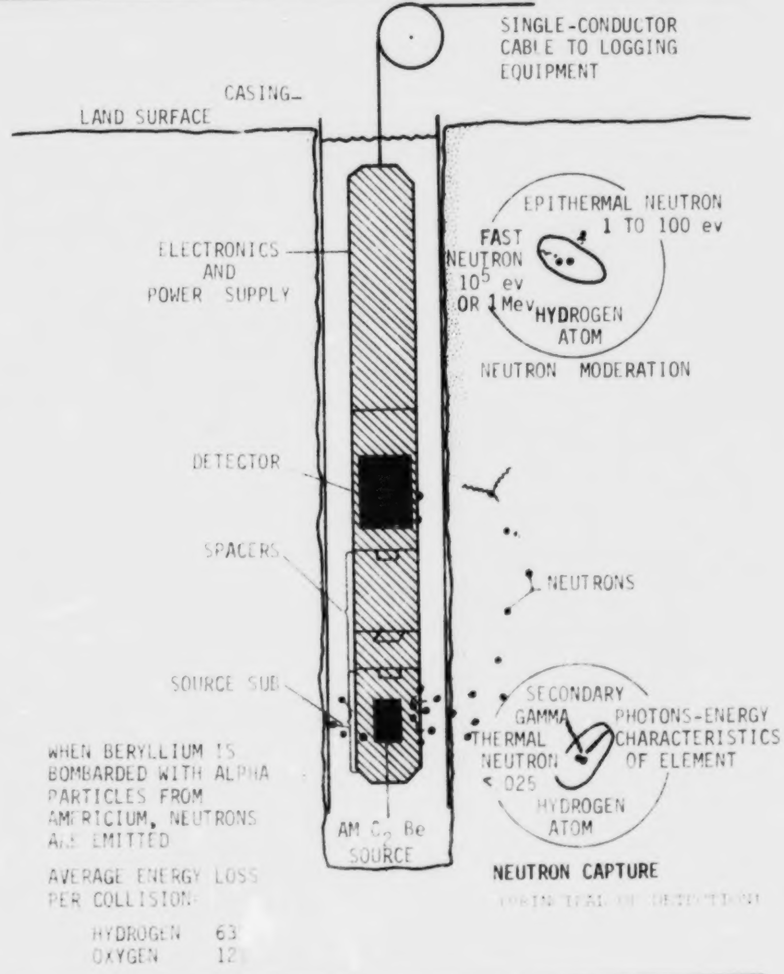

SINGLE-CONDUCTOR CABLE TO LOGGING EQUIPMENT
CASING
LAND SURFACE
EPITHERMAL NEUTRON 1 TO 100 ev
FAST NEUTRON 10^5 ev OR 1 Mev
HYDROGEN ATOM
NEUTRON MODERATION
ELECTRONICS AND POWER SUPPLY
DETECTOR
SPACERS
NEUTRONS
SOURCE SUB
SECONDARY GAMMA
PHOTONS-ENERGY CHARACTERISTICS OF ELEMENT
THERMAL NEUTRON < 025
HYDROGEN ATOM
NEUTRON CAPTURE
WHEN BERYLLIUM IS BOMBARDED WITH ALPHA PARTICLES FROM AMERICIUM, NEUTRONS ARE EMITTED
AM C_2 Be SOURCE
AVERAGE ENERGY LOSS PER COLLISION:
HYDROGEN 63
OXYGEN 12

Figure 4. Equipment and principles of neutron moisture logging (from Keys and MacCary, 1971)

only small-diameter (e.g., 2 inch) access wells. Larger units with multicurie sources are used to log wells with larger casing. The most common detector of slow neutrons is lithium-enriched BF$_3$. The relative positions of source and detector in the down-hole tool are critical (see van Bavel, 1963). The pulse emitted during capture is amplified by a preamplifier within the down-hole tool and sent through the cable to an above-ground meter for counting (see Figure 4), or recording the output.

Access tubes for neutron moisture logging are usually constructed of seamless steel or aluminum tubes. Wells cased with plastic (PVC, ABS, etc.) may cause difficulty in that hydrogen or chlorine atoms present in the casing may moderate the thermal neutrons, interfering with soil moisture evaluation. Aluminum-cased wells may deteriorate in saline soils.

The inside diameter of wells should be as close as possible to the outside diameter of the probe. Work by Ralston (1967) showed that for a 100-mc Am-Be source in a 1.5-inch O.D. tool, water content could not be accurately evaluated in wells greater than 4 inches. Brakensiek, Osborn and Rawls (1979) recommend that the radial air gap should not exceed 0.02 inch. Thus, for large-diameter casing it is advisable to use loggers with sources in the multicurie range.

Careful installation of wells is of critical importance in ensuring the accuracy of the method. As pointed out by McGowan and Williams (1980), a neutron probe has a large sphere of influence but readings are particularly sensitive within a narrow region of soil, perhaps only a few millimeters in thickness, immediately surrounding the well. Unfortunately, this region is most susceptible to damage during construction. Unless the well is tightly installed, cavities may promote the preferential flow of water along the casing wall, leading to erroneous results.

As with all indirect methods, the neutron attenuation method requires the preparation of calibration curves relating water content to the output signal of surface components (e.g., count rate). Generally a curve is supplied with each unit, but for the sake of accuracy, curves should be prepared for on-site materials. Visvalingam and Tandy (1972), and Brakensiek, Osborn and Rawls (1979) discussed calibration techniques in detail. Briefly, in laboratory calibration the sample must be of homogeneous water content, structure, and density, and of large enough volume to encompass the sphere of influence of the source. A series of samples with distinct water contents must be prepared. The samples and a central access tube are placed in a suitable container. Logs are obtained for each sample. Output values of the neutron logger are then plotted against water content values. Inasmuch as disturbed samples are used, the calibration curve may or may not represent field conditions.

One approach for field calibration is to collect solids samples during installation of access wells and determine water contents gravimetrically. Immediately after installing each well, neutron moisture logs are obtained. The logger output values for each depth are plotted against the corresponding water content values.

Advantages and disadvantages of the neutron method are discussed in detail in the catalog of methods at the end of this article. One advantage which must be emphasized is that neutron logs may show the presence of perched ground water in the vadose zone, thus assisting in positioning monitoring wells for sampling water in transit through the vadose zone. Another advantage relative to vadose zone monitoring near ponds, lagoons, and landfills is that access wells can be installed on the periphery of such facilities. Moisture logs from such wells may detect the horizontal spread of waste water. The sequence of logs shown on Figure 5, obtained in an access well near a recharge pit, exemplify perched ground-water development. This example shows that neutron moisture logging in peripheral access wells could be used as an alter-

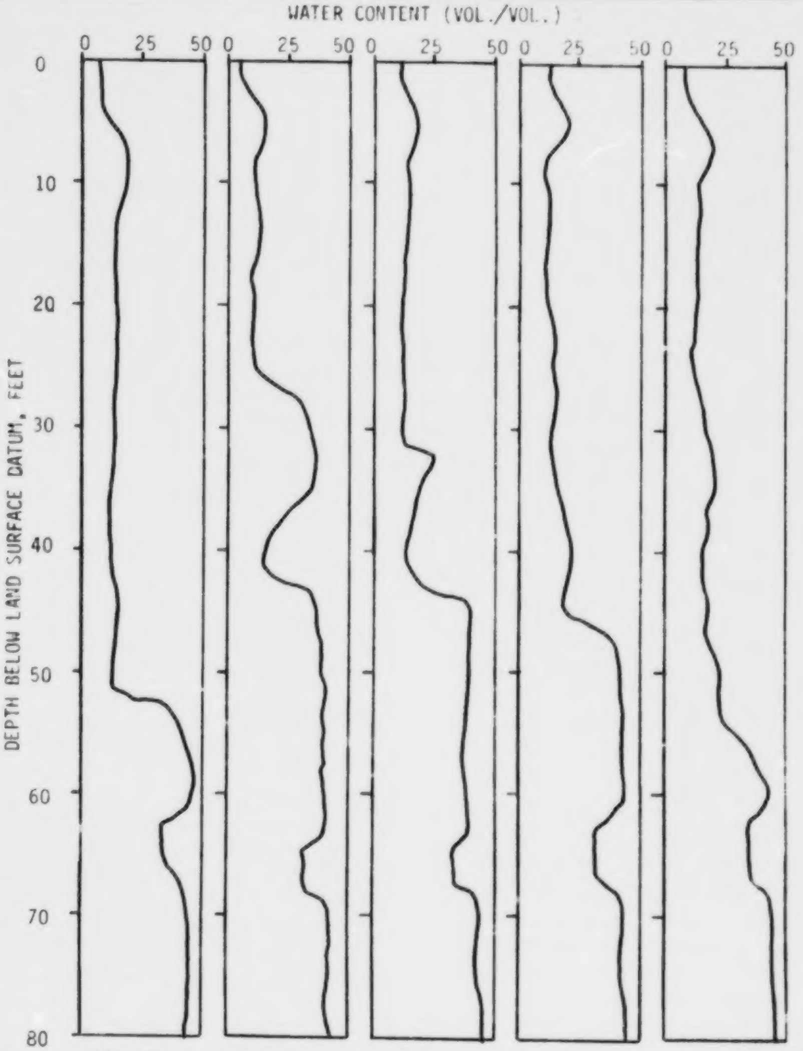


Figure 5. Sequence of water content profiles obtained in the vadose zone during pit recharge studies near Tucson, Arizona

native monitoring technique for impoundments in lieu of sampling methods excluded by the Hazardous Waste and Consolidated Permit Regulations. An important disadvantage of neutron moisture logging is that water may move through a specific subsurface horizon without causing a change in storage (Hillel, 1980). In this case, a neutron log would not manifest water movement.

Catalog of Methods for Monitoring Water Content in the Vadose Zone

Table 1 constitutes a catalog of common methods which have been or could be used for monitoring water content in the vadose zone. The emphasis is on methods possibly useful in deeper regions. Remote sensing approaches are excluded because their focus is on shallow regions. For an excellent review of remote sensing methods and others, including those in Table 1, the reader should consult Schmugge, Jackson and McKim (1980).

Biographical Sketch

L. G. Wilson received his B.S. in agriculture from the University of British Columbia in 1951, and his M.S. in irrigation science and Ph.D. in soil physics from the University of California in 1957 and 1962 respectively. From 1956 to 1960 he served as assistant specialist in irrigation at the University of California. Since 1962, he has been employed as a hydrologist at the Water Resources Research Center, University of Arizona, Tucson, Arizona. Wilson has written journal articles and technical articles on vadose zone monitoring, most notably an EPA-sponsored report entitled "Monitoring in the Vadose Zone: A Review of Technical Elements and Methods" (1980). He has also been involved in a number of studies on vadose zone and saturated zone monitoring, and has designed monitoring networks for recharge pits, oxidation ponds, landfills, irrigation return-flow systems, and both coal strip mining and oil shale processing operations.

References

Ayers, R.S., and R.L. Branson, editors. 1973. Nitrates in the Upper Santa Ana River Basin in relation to ground-water pollution. California Agricultural Experiment Station Bulletin 861.

Bianchi, W.C. 1967. Measuring soil moisture tension changes. Agricultural Engineering v. 43, no. 7, pp. 398, 399, 404.

Bouwer, H. 1978. Ground Water Hydrology. McGraw-Hill Book Co., New York, N.Y.

Bouwer, H., and R.D. Jackson. 1974. Determining soil properties. In: Drainage for Agriculture. J. van Schilfgaarde, editor. Agronomy, no. 17, published by the American Society of Agronomy, Madison, Wis.

Brakensiek, D.L., H.B. Osborn, and W.J. Rawls. 1979. Field manual for research in agricultural hydrology. Agricultural Handbook, no. 224, published by the U.S. Department of Agriculture, Science and Education Administration.

Briscoe, R. 1979. Effective use of thermocouple psychrometers in the measurement of water. Paper presented at the annual meeting of the Agronomy Society of America, Fort Collins, Colo., August 3, 1979.

Campbell, E.C., G.S. Campbell, and W.K. Barlow. 1973. A dewpoint hygrometer for water potential measurement. Agricultural Meteorology, v. 12, pp. 113-121.

Enfield, C.G., J.J.C. Hsieh, and A.W. Warrick. 1973. Evaluation of water flux above a deep water table using thermocouple psychrometers. Soil Science Society of America Proceedings, v. 37, no. 6, pp. 968-970.

Evans, D.D., T.W. Sammis, and A.W. Warrick. 1976. Transient movement of water and solutes in unsaturated soil systems. Project completion report, Department of Hydrology and Water Resources, The University of Arizona, Tucson.

Freeze, R.A., and J.A. Cherry. 1979. Ground Water, published by Prentice-Hall Inc., Englewood Cliffs, N.J.

Gairon, S. and S. Hadas. 1973. Measurement of the water status in soils. In Arid Zone Irrigation, Yaron B., E. Dantois and Y. Vaadia, editors. Published by Springer-Verlag, New York, N.Y., pp. 215-266.

Gardner, W.H. 1965. Water content. In Methods of Soil Analyses, Black, C.A., ed., Agronomy, no. 9, published by the American Society of Agronomy, Madison, Wis.

Hanks, R.J., and G.L. Ashcroft. 1980. Applied soil physics. Published by Springer-Verlag, Berlin, Heidelburg, New York.

Hillel, D. 1971. Soil and water, physical principles and processes. Published by Academic Press, New York, N.Y.

Hillel, D. 1980. Applications of soil physics. Published by the Academic Press, New York, London, Toronto, Sydney, San Francisco.

Holmes, J.W., S.A. Taylor, and S.J. Richards. 1967. In Irrigation of Agricultural Lands, Hagan, R.M., H.R. Haise, and T.W. Edminster, editors, Agronomy, no. 9, American Society of Agronomy, Madison, Wis., pp. 275-298.

Keys, W.S. 1967. The application of radiation logs to ground-water hydrology. In Symposium on Use of Isotopes in Hydrology. International Atomic Energy Agency, pp. 477-488.

Keys, W.S., and L.M. MacCary. 1971. Application of borehole geophysics to water resources investigations. In Techniques of Water Resources Investigations of The United States Geological Survey, Chapter E1, Book 2, Collection of Environmental Data.

McGowan, M., and J.B. Williams. 1980. The water balance of an agricultural catchment. I: Estimation of evaporation from soil water records. Journal of Soil Science, v. 31, pp. 217-230.

Mann, J.F. 1976. Wastewaters in the vadose zone of arid regions. Ground Water, v. 14, no. 6, pp. 367-372.

Merrill, S.D. and S.C. Rawlins. 1972. Field measurement of soil water potential with thermocouple psychrometers. Soil Science, v. 113, no. 2, pp. 102-109.

Oaksford, E.T. 1978. Water-manometer tensiometers installed and read from the land surface. Geotechnical Testing Journal, v. 1, no. 4, pp. 199-202.

Paetzold, R.F. 1979. Measurement of soil physical properties. In Infiltration Research Planning Workshop. Part 1. State of The Art Reports. Science and Education Administration, ARM-NC-4, U.S. Department of Agriculture.

Phene, C.J., G.J. Hoffman, and S.J. Rawlins. 1971a. Measuring soil matric potential in-situ by sensing heat dissipation within a porous body. I: Theory and sensor construction. Soil Science Society of America Proceedings, v. 35, pp. 27-33.

Phene, C.J., S.J. Rawlins, and G.J. Hoffman. 1971b. Measuring soil matric potential in-situ by sensing heat dissipation within a porous body. I: Experimental results. Soil Science Society of America Proceedings, v. 35, pp. 225-229.

Ralston, D.R. 1967. Influence of water well design on neutron logging. Unpublished M.S. Thesis, The University of Arizona.

Rawlins, S.L., and F.W. Dalton. 1967. Psychrometric measurement of soil water potential without precise temperature control. Soil Science Society of America Proceedings, v. 31, pp. 297-301.

Reginato, R.J., and R.D. Jackson. 1971. Field measurement of soil water content by gamma-ray transmission compensated for temperature fluctuations. Soil Science Society of America Proceedings, v. 35, pp. 529-533.

Reginato, R.S., and C.H.M. van Bavel. 1964. Soil water measurement with gamma attenuation. Soil Science Society of America Proceedings, v. 28, pp. 721-724.

Reynolds, S.G. 1970a. The gravimetric method of soil moisture determination. I: A study of equipment and methodological problems. Journal of Hydrology, v. 11, no. 3, pp. 258-273.

Reynolds, S.G. 1970b. The gravimetric method of soil moisture determination. II: Typical required sample sizes and methods of reducing variability. Journal of Hydrology, v. 11, no. 3, pp. 274-287.

Richards, L.A. 1965. Physical condition of water in soils. In: Methods of Soil Analysis, C.A. Black, editor, Agronomy, no. 9, published by the American Society of Agronomy, pp. 128-152.

Runnells, D.D. 1976. Wastewaters in the vadose zone of arid regions: geochemical interactions. Ground Water, v. 14, no. 6, pp. 374-385.

Schmugge, T.J., T.J. Jackson, and H.L. McKim. 1980. Survey of methods for soil moisture determination. Water Resources Research, v. 16, No. 6, pp. 961-979.

U.S. Environmental Protection Agency. 1980. Hazardous waste and consolidated permit regulations. Federal Register, v. 45, no. 98, book 2, pp. 33063-33588.

van Bavel, C.H.M. 1963. Neutron scattering of soil moisture: development and current status. Proceedings International Symposium Humidity and Moisture, Washington, D.C., pp. 171-184.

Visvalingam, M., and J.D. Tandy. 1972. The neutron method for measuring soil moisture content. Journal of Soil Science, v. 23, no. 4, pp. 499-510.

Watson, K.K. 1967. A recording field tensiometer with rapid response characteristics. Journal of Hydrology, v. 5, pp. 33-39.

Wilson, L.G. 1980. Monitoring in the vadose zone: a review of technical elements and methods. U.S. Environmental Protection Agency. EPA-600/7-80-134. 168 pp.

Wilson, L.G., and K.J. DeCook. 1968. Field observations on changes in the subsurface water regime during influent seepage in the Santa Cruz River. Water Resources Research, v. 4, no. 6, pp. 1219-1234.

Winograd, I.J. 1974. Radioactive waste storage in the arid zone. EOS, Transactions American Geophysical Union, v. 55, no. 10, pp. 884-894.

Table 1. Catalog of Methods for Monitoring Water Content in the Vadose Zone

Method	Principle	Advantages	Disadvantages	References
1. Gravimetric a. Oven drying	Core samples are obtained from the vadose zone using tube samplers for shallow depths and hollow stem auger plus core sampling for greater depths. A core sample is weighed, oven dried at 105 C for 24 hours, and reweighed. The water content is determined by difference in weight. Results expressed on a dry weight or volume basis. The difference in water content values of successive samples represents change in storage.	1. A direct method. 2. The most accurate of available methods. 3. Simple.	1. A large number of replicate samples are required for each depth increment (necessitating several holes) to account for spatial variability of water holding properties. 2. Expensive if large numbers of samples are required. 3. A destructive method — i.e., additional measurements cannot be obtained at the same sites.	Gardner (1965), Hillel (1971), Schmugge, Jackson and McKim (1980), Reynolds (1970a, 1970b), Brakensiek, Osborn and Rawls (1979).
b. Carbide method	A field method. Solids samples are placed in a container with calcium carbide. The calcium carbide reacts with water, releasing a gas. The gas pressure, registered on a gage, is converted into water content on a dry weight basis.	1. More rapid than oven drying. 2. Initial capital investment is lower than for oven drying.	1. May not be as accurate as oven drying. 2. Other disadvantages are the same as for oven drying.	

Method	Principle	Advantages	Disadvantages	References
2. Neutron moisture logging (neutron scatter method)	A source of high energy neutrons (e.g., amerecium-beryllium) in a down-hole tool is lowered into an access well. Water in the vadose zone slows down the fast neutrons, which are captured by a detector in the tool. Counts are measured by a surface scaler, ratemeter, or recorder. Counts are converted into volumetric water content by an appropriate calibration relationship. Successive readings show temporal changes in water storage at successive depths.	1. Rapid. 2. An in situ method. 3. Can be conducted in cased or uncased holes (for safety in unstable material should install casing). 4. Can be interfaced with portable data collection system. 5. Successive readings are obtained in the same profile at the same field location. 6. Can be used to locate perched ground-water zones, i.e., valuable for positioning monitoring wells for sampling perched ground water.	1. Expensive, requiring the purchase or lease of equipment. 2. Water content is measured in a sphere. Cannot relate results exactly to a specific depth. 3. Fast neutrons are moderated by other constituents besides hydrogen in water, e.g., chlorine or boron. Accuracy may be affected. 4. During installing of access wells, cracks or cavities may be formed causing leakage along the casing wall. 5. An indirect method requiring calibration. Calibration is a difficult procedure. 6. Accurate readings are not possible within 6 in. of soil surface. 7. Cannot be used to infer water movement in regions where storage changes do not occur.	Holmes, Taylor and Richards (1967), van Bavel (1963), Keys and MacCary (1971), McGowan and Williams (1980), Schmugge, Jackson and McKim (1980), Wilson (1980), Hillel (1971), Brakensiek, Osborn and Rawls (1979), Visvalingum and Tandy (1972).
3. Gamma ray attenuation a. Transmission method	Two parallel wells installed at precise distances apart are required. A probe with a gamma photon source (e.g. cesium 137) is lowered in one well. A second probe with a detector (e.g. sodium iodide scintillation crystal) is lowered at the same rate in the second well. Accessories include a high-voltage supply, amplifier, scaler, timer, spectrum analyzer, pulse height analyzer and photomultiplier tube. The degree to which a beam of monoenergetic gamma rays is attenuated depends on the bulk density and water content. Assuming that the bulk density remains constant, changes between readings reflects changes in water content.	1. A rapid, in-situ method. 2. Water content is obtained in a narrow beam—depth-wise measurement can be obtained as close as one inch apart. 3. Measurements can be obtained within one inch of surface. 4. Nondestructive and successive measurements are obtained at same locations. 5. Can be interfaced with portable data collection system.	1. Limited to shallow depths because of difficulties in installing precisely parallel wells, particularly in rocky material. 2. Instabilities in count rate may occur. 3. Expensive. 4. Changes in bulk density in shrinking-swelling material affects accuracy of water content readings. 5. Variations in water content and bulk density occur in stratified soils. 6. Care must be taken in handling radioactive source.	Brakensiek, Osborn and Rawls (1979), Gardner (1965), Bouwer and Jackson (1974), Reginato and van Bavel (1964), Reginato and Jackson (1971), Schmugge, Jackson and McKim (1980).
b. Scattering method	A single probe is used, containing a gamma source and a detector, separated by a lead shield. Gamma rays beamed into the surrounding media are absorbed by the solid media and water. Back scattered rays are detected and measured. Knowing the dry bulk density of the media, the water content can be calculated. Requires empirical calibration curves.	1. Rapid. 2. Nondestructive, with successive measurements obtained at same depth. 3. In contrast to the transmission method only one access well is required. Readings can be obtained at great depth in vadose zone.	1. Requires a source of higher strength than transmission method. 2. Not as accurate as transmission method because water content measured in sphere and not a beam. 3. Expensive. 4. Changes in bulk density in shrinking, swelling material changes calibrations.	Keys and MacCary (1971), Brakensiek, Osborn and Rawls (1979), Paetzold (1979).

Method	Principle	Advantages	Disadvantages	References
4. Tensiometers	A tensiometer consists of a porous ceramic cup cemented to rigid plastic tube, containing small diameter tubing leading to a surface reservoir of mercury. Alternate version uses strain gage transducer in lieu of mercury manometer. The body tubing is filled with water. Pores in cup form continuum with pores in exterior medium. Water moves into or out of body tube until equilibrium is reached. Measured water pressure reflects corresponding water pressure in medium. By using appropriate soil water characteristic curve, pressure can be related to water content.	1. Provide continuous, in place measurements of water content. 2. Successive measurements are obtained. 3. Inexpensive and simple. 4. Transducer units respond fairly rapidly to water content changes.	1. Units fail at the air entry value of the ceramic cup, generally about -0.8 atmospheres. 2. Results are subject to hysteresis, that is, different results are obtained for wetting vs. drying media. 3. If proper contact is not made between cup and media units will not operate properly. 4. Sensitive to temperature changes. 5. Difficult to install at great depth in vadose zone.	Brakensiek, Osborn and Rawls (1979), Holmes, Taylor and Richards (1967), Bianchi (1967), Gairon and Hadas (1973), Schmugge, Jackson and McKim (1980), Wilson (1980), Oaksford (1978).
5. Electrical resistance blocks	Blocks consist of electrodes embedded in porous material (plaster of paris, nylon, cloth, fiberglass). Water content of blocks change with water content of soil. Electrical properties of blocks change with changing water content. Electrical properties are measured using a meter. Calibration curves must be obtained.	1. Can be interfaced with portable data collection system. 2. Can be used at soil water pressures less than -0.8 atmospheres. 3. Gypsum blocks are inexpensive. 4. Precision is good.	1. Subject to hysteresis. 2. May be difficult to install at great depth in vadose zone and maintain good contact. 3. Requires calibration for each textural type in profile. 4. Lack of insensitivity in wet range. 5. Sensitivity to soil salinity (except gypsum blocks). 6. Gypsum blocks deteriorate badly in certain media. 7. Calibration curves of some units shift with time. 8. Time lag in response.	Brakensiek, Osborn and Rawls (1979), Holmes, Taylor and Richards (1967), Phene, Hoffman and Rawlins (1971), Schmugge, Jackson and McKim (1980), Gairon and Hadas (1973).
6. Thermocouple psychrometers/hygrometers	A psychrometer unit consists of a porous bulb with a chamber in which the relative humidity of the exterior media is sampled, a sensitive thermocouple, a heat sink, reference electrode, and electrical circuitry. The unit operates on the principle that a relationship exists between soil water potential and relative humidity. Two types are available, the wet bulb type and the dew point type. Both types rely on cooling of the thermocouple junction by the Peltier effect. In the wet bulb type, when the temperature of the junction is reduced below the dew point, cooling is discontinued. As condensed water evaporates, the temperature increases to ambient. Signal from the junction at the temperature plateau is proportional to relative humidity. In the dew point type, the temperature at junction is held constant at dew point. The thermocouple signal corresponds to dew point depression.	1. In-situ pressure measurements are possible down to -50 atmospheres, permitting the determination of water contents in the very dry range. 2. Permits continuous recording of pressures (and water contents) at the same depth. 3. Can be interfaced with portable or remote data collection systems. 4. Some units have been installed to great depth (down to 300 feet).	1. Results are subject to hysteresis. 2. Good contact between bulb and surrounding media may be difficult to obtain. 3. Provide point measurements only. 4. May be difficult to obtain accurate calibration curves for deep regions of the vadose zone. 5. Fragile, requiring great care in installation.	Rawlins and Dalton (1967), Merrill and Rawlins (1972), Enfield, Hsieh and Warrick (1973), Schmugge, Jackson and McKim (1980), Hanks and Ashcroft (1980), Briscoe (1979), Campbell, Campbell and Barlow (1973).

	and thus to the relative humidity. Different methods are required for the two types. The dew point method is more accurate. Calibration curves relating relative humidity to water potential are required. Water potential and water content are related through a characteristic curve for each material.			
7. Heat dissipation sensor	Heat dissipation sensors operate on the principle that the temperature gradient to dissipate a given amount of heat in a porous medium of low conductivity is related to water content. In practice, the water content of a soil can be measured by applying a heat source at a central point within the sensor and measuring the temperature rise at that point. Calibration curves of matric potential vs. temperature difference are obtained using a pressure plate apparatus with soils from the site. The matric potential is related to water content by preparing a water characteristic curve. Commercial sensors consist of a miniature heater, temperature sensors and circuitry, embedded in a cylindrical porous ceramic block within a small diameter PVC tube, and a lead cable.	1. Simple. 2. May be interfaced with a data acquisition system for remote collection of data. 3. Measurements are independent of salt content of soil. 4. Calibration appears to remain constant. 5. Can be used to measure soil temperature as well as matric potential. 6. Useful for measuring water contents in the dry range.	1. Subject to hysteresis in the water characteristic. 2. Calibration is required for each change in texture. 3. May be difficult to install at depth in the vadose zone and maintain good contact between the sensor and medium.	Phene, Hoffman and Rawlins (1971a). Phene, Rawlins and Hoffman (1971b). Schmugge, Jackson and McKim (1980).

MONITORING OF LEACHATE MIGRATION IN THE UNSATURATED ZONE IN THE VICINITY OF SANITARY LANDFILLS

By Thomas M. Johnson,
Keros Cartwright and
Rudolph M. Schuller

Introduction

Guidelines for the siting and design of landfills have generally required that waste disposal take place above the zone of saturation. Regulatory agencies usually require that small-diameter observation wells be used to monitor changes in the quality of ground water in the vicinity of sanitary landfills. Unsaturated soils often will not yield water to wells because moisture is held in the soil under surface tension at less-than-atmospheric pressures. Thus, monitoring of the unsaturated zone beneath landfills has been neglected. By the time ground-water contamination is detected via monitoring wells, corrective action is expensive and time-consuming. Because the unsaturated zone beneath a landfill represents a buffer zone between refuse and potential aquifers, monitoring of this zone would allow for early detection of leachate migration. In addition, monitoring the changes in water quality in the unsaturated zone provides information on the attenuation capacities of unsaturated materials and the rates of ground water movement, both of which are very difficult to estimate.

The usefulness and reliability of methods for monitoring water quality in unsaturated materials have been demonstrated, but their practical application in pollution control programs has been limited. Also, although several types of remedial measures are applicable to leachate migration in the saturated zone, corrective actions for leachate migration in the unsaturated zone are severely limited.

Pressure-Vacuum Lysimeters

One method of sampling soil moisture is with porous ceramic cups similar to those used in tensiometers. Soil sampling requires repeated drilling for extended analyses; the use of soil-moisture samplers, called suction, or pressure-vacuum, lysimeters, allows repeated sample collection from a single installation.

The original design of the pressure-vacuum lysimeter permitted sampling only at shallow depths. This lysimeter was modified during several studies in Pennsylvania concerning the effects of the infiltration of waste water and the migration of landfill leachate (Parizek and Lane, 1970; Apgar and Langmuir, 1971). Sampling at greater depths was achieved by the use of two access tubes and a ceramic cup attached to a sample accumulation chamber of PVC pipe which held approximately one liter (Figure 1). To collect a soil-moisture sample, a partial vacuum is applied to the sampler through one of the access tubes, which is then clamped shut. Soil moisture held under tension moves into the sampler under the potential gradient created by the vacuum, and water generally accumulates. When the vacuum is released and pressure is applied to one access tube, the accumulated water sample is forced to the surface through the other access tube.

In studies of landfills in Pennsylvania, several pressure-vacuum lysimeters were installed in individual boreholes at depths to 16.8 m (55 ft.) The movement and attenuation of leachate were monitored by standard indicators of water quality. Apgar and Langmuir (1971) concluded that improperly designed landfills located above the water table in permeable materials can result in serious ground water pollution.

Wood (1973) described a modification of the lysimeters used by Apgar and Langmuir that permitted the sampling of water from any depth within the unsaturated zone. The use of a check valve prevents pressurization of the porous cups; in deeply placed lysimeters, pressures in excess of one atmosphere in the sampler would force the water sample back through the cup rather than up to the surface.

More recent applications have included the use of suction lysimeters to monitor the quality of water in the unsaturated zone at depths to 33 m (108 ft.) below artificial recharge sites in Texas (Wood, 1978). In Wisconsin, three operating landfills were instrumented with suction lysimeters to monitor leachate migration and attenuation in the unsaturated zone (Gerhardt, 1977).

The validity of soil-moisture samples collected by suction lysimeters with ceramic cups is extremely important if the results are to be used for regulatory purposes. Because some studies have indicated that ceramic cups can alter the chemical composition of samples, such samples may not be representative of actual soil-moisture quality.

Substantial bias and variability have been found in soil-water samples of NO_3-N, PO_4-P, Na, K and Ca by previous investigators (Hansen and Harris, 1975; Grover and Lamborn, 1970). Factors including intake rate, plugging of ceramic-cup pores, and sorption and screening of some ions produced as much as 60 percent range in sample concentrations. Ceramic cups contributed excessive amounts of Ca, Na and K to samples of soil moisture having low solute concentrations and significant amounts of P were adsorbed on the ceramic cups. Rinsing of the cups with dilute HCl reportedly reduced Na and K contamination and P adsorption to acceptable levels.

Dazzo and Rothwell (1974) found that large-scale screening and adsorption of bacteria by ceramic cups with a pore size of 3 to 8 μm invalidated their use for the analysis of fecal coliform. The effective pore diameter of the porous ceramic cup used in the lysimeters is approximately 1μm, which may allow colloidal particles to pass through. Wolf (1967) found that new ceramic cups yield several milligrams per liter of Ca, Mg, Na, HCO_3 and SiO_2, after cleaning with dilute HCl. Wagner (1962) found no measurable sorption of NO_3-N. Levin and Jackson (1977) indicated that concentrations of Ca, Mg and PO_4-P in soil solutions were unaltered by ceramic cup samplers and that clogging was not a problem; however, their suitability for NO_3-N was questionable.

Each of the previous studies utilized solutions of relatively low solute concentrations where the effects of sampling errors and sample variability are significant. Despite these problems, soil-water samplers are now commonly used for monitoring highly contaminated soil solutions such as landfill leachate. The ability of these samplers to obtain samples that are truly representative of highly concentrated leachate has not been examined.

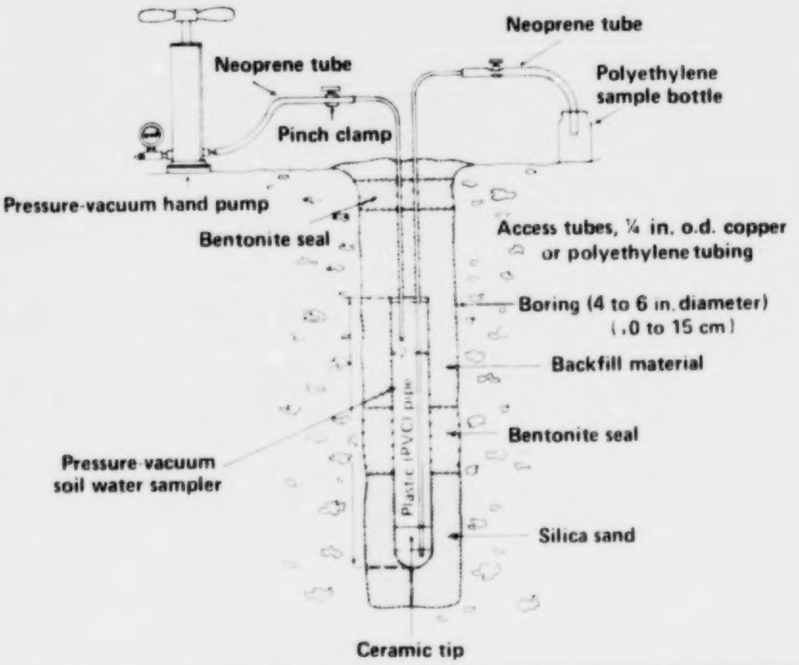

Neoprene tube
Neoprene tube
Polyethylene sample bottle
Pinch clamp
Pressure-vacuum hand pump
Bentonite seal
Access tubes, ¼ in. o.d. copper or polyethylene tubing
Boring (4 to 6 in. diameter) (10 to 15 cm)
Backfill material
Plastic (PVC) pipe
Bentonite seal
Pressure-vacuum soil water sampler
Silica sand
Ceramic tip

Figure 1 Typical installation of pressure-vacuum lysimeter.

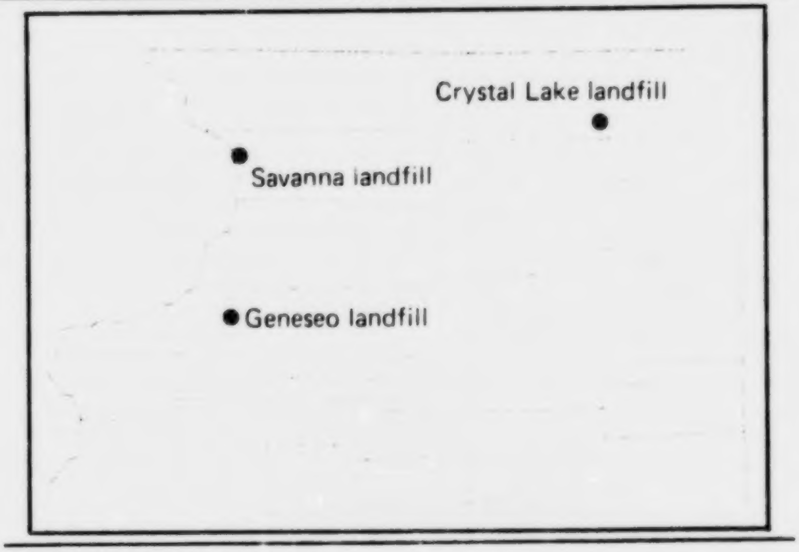


Figure 2 Locations of study sites.

Description of the Study

A project was undertaken to investigate the movement and quality of leachate in the unsaturated zone beneath several landfills in Illinois and to evaluate methods of monitoring soil-moisture and ground-water conditions (Johnson and Cartwright, 1980). Three landfills (Figure 2) representing different geologic environments and disposal practices were studied. Contaminant migration was monitored by means of observation wells and soil-water samplers (pressure-vacuum lysimeters) and by chemical analysis of soil samples. In addition, a laboratory study was conducted in a preliminary attempt to determine if there are any factors that significantly affect the validity of samples of highly contaminated landfill leachate collected with pressure-vacuum lysimeters.

The pressure-vacuum lysimeters used in this study were obtained from Soil Moisture Equipment Corp. (Model 1920), Santa Barbara, California.* A typical lysimeter installation is illustrated in Figure 1. The lysimeter device, which holds one liter of water, consists of a plastic cylinder, 64 cm (25 in.) long and 5 cm (2 in.) in diameter, having a porous ceramic cup 6 cm (2.5 in.) long. An envelope of silt-sized, powdered quartz is typically emplaced around the ceramic cup to establish uniform continuity with surrounding soils. The suction lysimeters were placed at varying depths beneath each site, one to three lysimeters in each borehole. Bentonite clay was used to seal the borehole between units.

Sample collection periods ranged from three days to three months. After a sample had accumulated, a hand pump was used to apply pressure to one access tube to force the water sample to the surface through the discharge tube that extends to the base of the sampler. (Gas pressure [argon, nitrogen, CO_2, etc.] could be used for sample collection; however, the effects of these gases on sample integrity are unknown.)

Soil moisture and ground-water samples were collected and analyzed for Na, Ca, Mg, Zn, K, Fe, Pb, Hg, Cl, SO_4, NO_3-N, NH_4-N; hardness; pH; total dissolved solids (TDS); and chemical oxygen demand (COD). Field measurements of pH, Eh, specific conductivity, and alkalinity were also made. In addition to water quality analyses, soil samples collected from the unsaturated zone beneath each landfill were analyzed to determine their attenuation capabilities and the extent of their contamination by the infiltrating leachate.

In the laboratory study, one new sampler that was not cleaned (designated as NC in subsequent figures) and one sampler that was cleaned with dilute hydrochloric acid and rinsed with distilled water (designated as C) were placed in separate plastic carboys, each containing approximately 15 liters of leachate. A constant flow of carbon dioxide gas was bubbled over the leachate in an effort to maintain reducing conditions and a constant pH.

The leachate was collected from a monitoring well (BL-2) finished in the refuse at the Blackwell Forest Preserve landfill in Du Page County (Hughes, Landon and Farvolden, 1971). This leachate was chosen because of its relatively high concentrations of constituents and ready availability of large volumes. In order to measure the validity of the lysimeters for sampling trace metals, each leachate was "spiked" with an additional 2.5 mg/l of Pb, Hg and Cu. A vacuum of 55 centibars (413 mm Hg) was initially placed on each sampler using a portable hand pump.

Duplicate samples of raw leachate were collected from each carboy for 10 days using air pressure applied with a hand pump. One sample (denoted AFNC or AFC) was immediately acidified to a pH <2.0 with concentrated nitric acid and, after centrifuging, was filtered through a 0.7 μm pore-size membrane filter. A similar sample (denoted FANC or FAC) was centifuged, filtered and then acidified prior to storage at 4 C. The large amount of suspended particulate matter in the raw leachate required that the samples be centrifuged prior to filtration. Samples of raw leachate were filtered through a 0.7 μm pore-size membrane filter in an attempt to simulate the pore diameter of the porous ceramic cups. The samples were analyzed for Ca, Mg, Na, K, Fe, Zn, Pb, Hg, Cu and Cl. The mean standard deviation and statistical analyses of the differences in mean concentrations (Student's t-test) and a comparison of the concentration variances (F-statistic) were determined for lysimeter samples (NC, C) and leachate (FANC, AFNC, AFC, FAC) in an effort to determine the effects on sample integrity of the leachate passing into the porous cup.

Results of the Field Study

Savanna Landfill

Located in an abandoned sand pit in the Mississippi River Lowland (Figure 2), the Savanna landfill is underlain by silty sand to coarse sand and gravel. Water levels generally range from 3 to 4 m (10 to 13 ft.) below the base of the refuse. Although the landfill has been operating since 1967, there has been no evidence of contamination in the monitoring wells or in the Plum River immediately south of the site. However, soil moisture both below and adjacent to the refuse is contaminated.

The lysimeters installed at the site yielded samples of soil moisture with an average volume of 300 ml. Analysis of these samples indicated that the migration of landfill leachate through the unsaturated zone was occurring beneath the site. Figure 3 shows a decrease of Cl in soil moisture and ground water with distance from the refuse. Similar patterns were observed for several other parameters, including COD, K, Na and Fe. Dilution and dispersion may be partly responsible for this apparent attenuation. Continued production of leachate by the landfill was indicated by the lack of reduction in the levels of chloride and COD in the soil moisture. Levels of most parameters are reduced to nearly background concentrations at depths of two to three m (6.5 to 10 ft.) below the refuse. Samples taken from the lysimeters also indicated that significant lateral migration of leachate occurred through the unsaturated zone adjacent to the refuse. Chemical analysis of soil samples below the landfill revealed that only Zn was present at levels slightly greater than background.

The Savanna landfill is situated in a permeable environment generally considered unsuitable for waste disposal. Samples of soil moisture indicate that the migration of highly contaminated leachate is occurring. The limited impact on water resources in the vicinity of the site is probably due to the limited quantities of leachate produced by the relatively small volume of refuse.

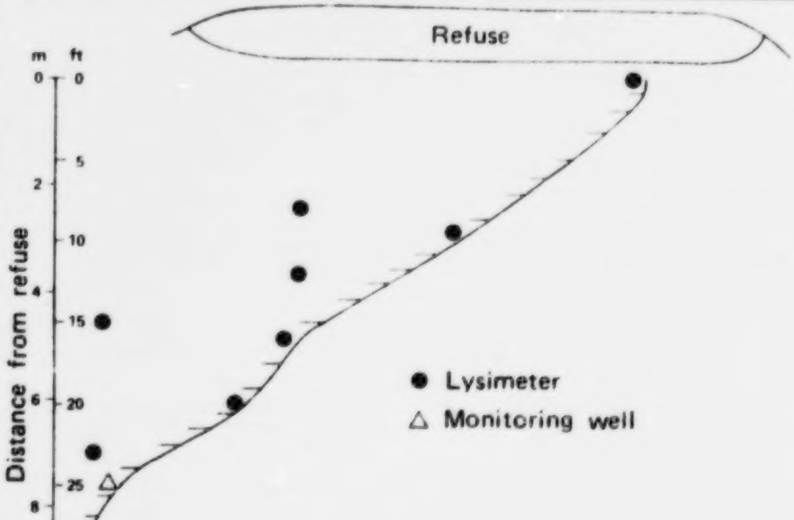


Figure 3 Concentration of chloride in water beneath the Savanna landfill March 17, 1977

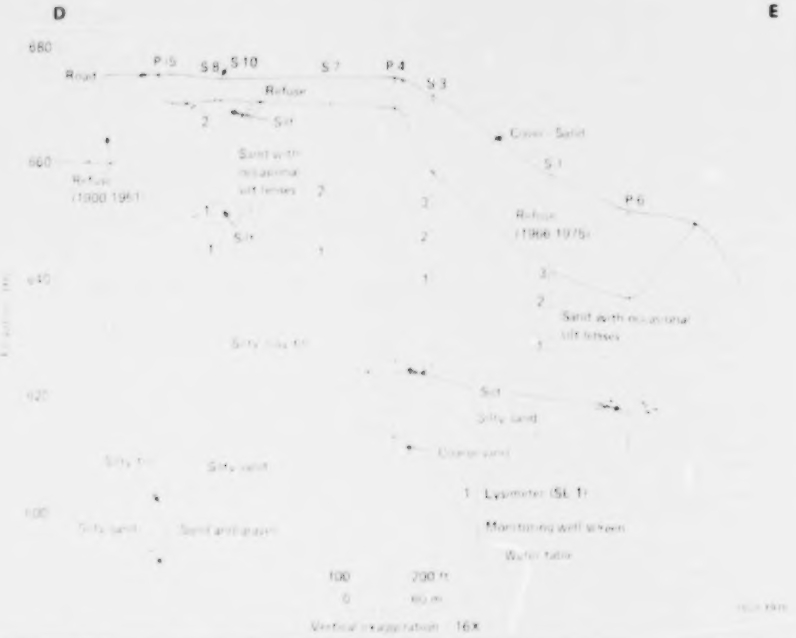


Figure 4 Cross section D-E of the Geneseo landfill and underlying materials showing a portion of the monitoring well-lysimeter network

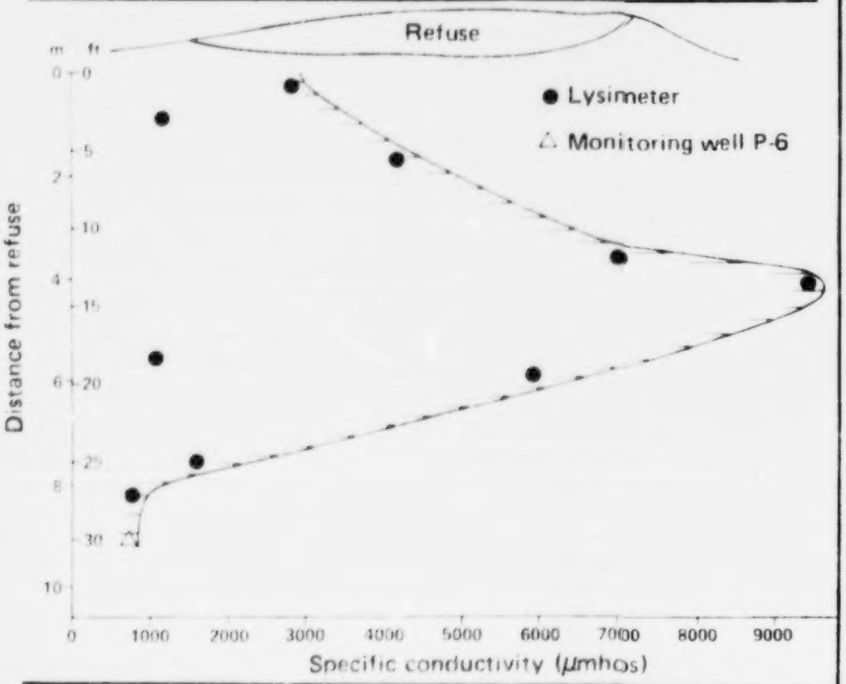


Figure 5 Specific conductivity of water beneath the Geneseo landfill, August 25, 1977

Geneseo Landfill

The Geneseo site is situated in a massive sand dune (Figure 2). The site originally began operation as an open-burning dump in the early 1900s and operated as a sanitary landfill from 1967 to 1975. The site was closed in 1975 following the suspected contamination of shallow municipal wells several hundred meters away. This study has disclosed the presence of relatively thick deposits of fine grained glacial till between the landfill and the underlying sand and gravel aquifer (Figure 4). Water levels range from 7 to 18 m (23 to 59 ft.) below the base of the refuse, and regional ground-water flow is toward the lowland to the north.

The average volume of each of the 56 samples obtained from the lysimeters at Geneseo was 316 ml. Highly contaminated soil moisture was collected beneath the landfill portion of the site. Less contaminated soil moisture was encountered beneath the remainder of the site. Sampling of monitoring wells and resampling of numerous municipal wells, however, has indicated virtually no leachate migration in the saturated zone.

Table 1 illustrates the range in quality of ground water and soil moisture at the site. Two processes apparently control the extent of contamination:

- the attenuation of leachate in the unsaturated zone
- an apparent decrease in the concentration of contaminants being released by the refuse. Many parameters, such as COD, were relatively stable over the two-year period encompassed by this study. Probably leachate has migrated through the unsaturated zone beneath the landfill at an average rate of 1 m/yr (3 ft/yr) since 1967. The actual rate of migration has probably been greater because much of the refuse was not emplaced until 1971 to 1975, and the sequence of materials beneath the site has caused highly variable rates in movement of soil moisture.

Soil-moisture samples from Geneseo indicated the expected effects of waste disposal in a permeable medium. The highly contaminated leachate migrating through the unsaturated zone beneath this site represents a significant threat to ground water. Maximum concentrations of most parameters, however, as illustrated by specific conductivity (Figure 5), are present at depths of 5 to 6 m (16 to 20 ft.) below the refuse. Below that depth, significant attenuation appears to have resulted from the presence of a layer of fine-grained glacial till that has restricted vertical migration and provided increased ion-exchange capacity. A significant amount of contaminated soil moisture probably flows laterally at the contact of the sand with the underlying glacial till, ultimately entering the ground-water system in the lowland to the north. However, no deterioration in water quality has yet been observed either in surface water or shallow ground water in the lowland north of the site. The pattern of leachate migration that was detected with the lysimeters also indicated that the suspected contamination of municipal wells was not due to leachate migration from the landfill.

Crystal Lake Landfill

The Crystal Lake landfill began operation in 1973 in an abandoned

TABLE 1. Composition of soil moisture and ground water at the Geneseo landfill.

	Leachate-soil moisture (mg/L)[a]	Uncontaminated soil moisture (mg/L)	Ground water (mg/L)
Ca	3,760	53	54
Fe	240	0.6	0.8
Hg	<.2 ppb	<.2 ppb	<.2 ppb
K	17	0.4	1.7
Mg	120	18	59
Na	1,260	18	31
Pb	<.01	<.01	<.01
Zn	0.17	0.01	0.06
Alk ($CaCO_3$)[b]	9,410	210	352
Cl	3,583	31	56
COD	8,898	6	457
NH_4-N	11	0.0	—
NO_3-N	7	0.2	1.5
pH[b]	7.05	6.05	7.70
SC[b]	7,000	525	750
SO_4	<.01	35	12
TDS	10,186	537	480

[a] Except Hg (ppb), pH, and SC (µmhos).

[b] Field measurement.

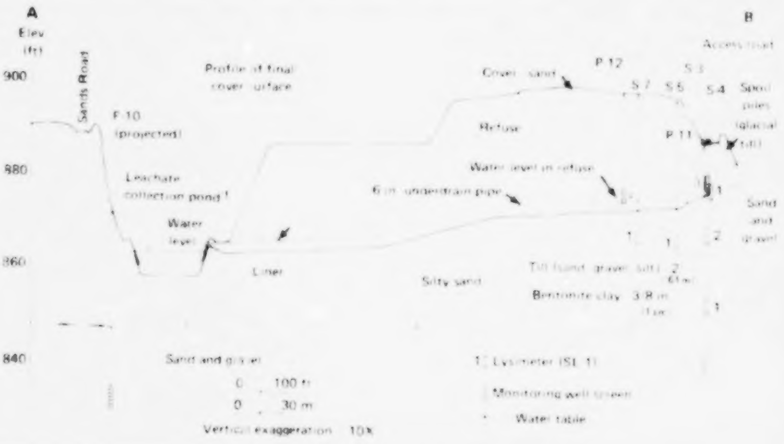


Figure 6 East-west cross section A-B of the Crystal Lake landfill showing the liner and relationships of monitoring devices to underlying materials. The base of the landfill slopes toward the west while the water table slopes gradually toward the east.

gravel pit (Figure 2). An artificial clay liner beneath the refuse consists of approximately 61 cm (2 ft.) of sandy glacial till from the adjacent spoil piles overlying a 1-cm (.5 in.) layer of bentonite clay spread over the underlying sand. Although the glacial till contained some silt and clay, it also contained much sand and many cobbles and boulders, which were not removed prior to installation of the liner. A leachate collection system consisting of underdrain pipe 15 cm (6 in.) in diameter was installed above the liner and led into a bentonite-lined lagoon at the west end of the fill area (Figure 6).

Water levels range from 7 to 10 m (23 to 33 ft.) below the refuse. The materials in this zone consist predominantly of very silty fine sand. The site is underlain by approximately 75 m (245 ft.) of glacial drift. Water-yielding sand and gravel units within the drift and the underlying Silurian dolomite are important aquifers in the area.

The liner and underdrain system were designed to collect leachate accumulating at the base of the refuse and divert it to the collection pond to be recovered and treated. However, during the eight years since refuse emplacement, the drain pipe has yielded water only during heavy rainfalls. Although three to six years may generally be required for refuse to reach field capacity, a well at the base of the refuse indicated a saturated zone of leachate at least 1 m (3 ft.) thick overlying the liner. The rapid response of the underdrain to precipitation events indicates that water is able to enter and flow through the refuse. Although a portion of the infiltration is diverted into the leachate collection system, the remainder is absorbed by the refuse or percolates through the liner.

The effectiveness of the clay liner could be evaluated only by measuring the quality of soil moisture migrating out of the landfill, because samples collected from wells at the site show no indication of contamination. The lysimeters installed beneath the clay liner clearly indicate the presence of contamination; however, the extremely low volumes of samples indicate low contents of soil moisture compared to that in similar materials adjacent to the refuse. The average volume of all the lysimeter samples was 330 ml. Contaminated soil moisture was found below the liner; comparison with the quality of leachate above the liner indicates some attenuation of many constituents migrating through the liner and the underlying silty sand (Table 2).

Lysimeters installed adjacent to the refuse also indicated the presence of leachate migration. Concentrations of most parameters in soil moisture decreased with distance from the refuse to approximate background levels at a distance of 7.6 m (25 ft.). Infiltration and attenuation processes adjacent to the refuse probably limit the effects of the lateral migration of leachate; Cl concentrations have been reduced by 95 percent at a distance of 3 m (10 ft.) from the refuse. However, in soil moisture beneath the refuse at a depth of 1.5 m (5 ft.), Cl concentrations have been reduced by only 10 percent of that in the raw leachate.

Results of the Laboratory Study

There have been no detailed experiments to evaluate the validity of samples of highly contaminated leachate collected with pressure-vacuum lysimeters. This preliminary laboratory study was designed to determine any significant changes that may be caused in the concentrations of several major constituents

TABLE 2. Composition of leachate, soil moisture, and ground water at the Crystal Lake landfill.

	P-12 Leachate (mg/L)[a]	S-7 Contaminated soil moisture (mg/L)	Uncontaminated soil moisture (mg/L)	P-8 Ground water (P-8) (mg/L)
Ca	6,200	1,720	96	47
Fe	350	85	0.0	0.4
Hg	< .2 ppb	< .2 ppb	< .2 ppb	<0.2 ppb
K	650	46	0.5	0.5
Mg	1,660	93	19	19
Na	1,030	220	7	2
Pb	0.4	< .01	< .01	< .01
Zn	46.5	0.24	0.03	0.6
Alk ($CaCO_3$)[b]	11,800	960	200	240
Cl	1,602	750	16	6
COD	32,000	440	5	4
NH_4-N		13	0.0	.
NO_3-N	7.9	38	0.3	3.9
pH[b]	5.65	7.40	6.35	7.48
SC[b]	12,000	3,400	700	520
SO_4	992	250	73	40
TDS	22,900	3,664	619	400

[a] Except Hg (ppb), pH, and SC (μmhos).

[b] Field measurement.

in samples of landfill leachate by collection in pressure-vacuum lysimeters.

In the laboratory study, the intake rate and hence, volume collected from each lysimeter, decreased steadily with time. The decrease in intake rate, or cup conductivity, is illustrated in Figure 7 as volume collected per unit time per atmosphere of applied vacuum. During the first five days, the intake rate of each sampler decreased rapidly; subsequently, the decrease, although steady, was at a much lower rate.

The effects of acid treatment of the ceramic cup may be reflected by the consistently larger sample volumes and intake rate of the acid cleaned lysimeter. This experiment indicated, however, that these effects appear to decrease with time when in contact with the raw leachate.

The decrease in yield of the samplers was probably the result of clogging of the small pores on the exterior of the ceramic cup by suspended particulates in the leachate. Microbial growth also may have contributed to clogging of pores. This decrease in yield, however, was not observed during field studies using identical lysimeters. Sample volume remained relatively constant from samplers at each of the three landfill sites. The suspended particulate matter primarily responsible for clogging of the ceramic cups is easily removed from leachate during migration through sediments beneath a landfill. Although microbial growth and plugging by colloids may still occur following migration through soils, these did not markedly affect the field performance of lysimeters during this study.

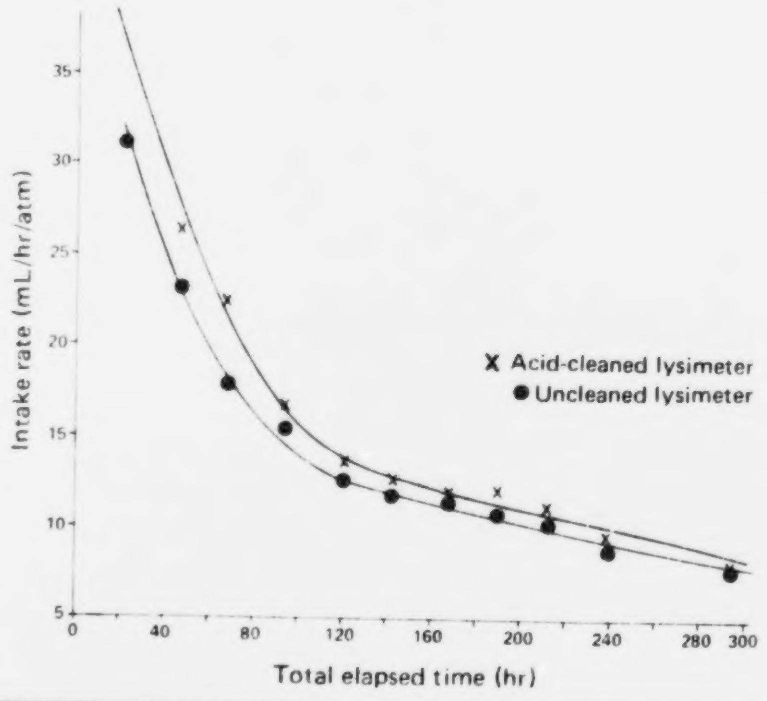


Figure 7. Intake rate (cup conductivity) of acid-cleaned and uncleaned lysimeters in landfill leachate versus elapsed time.

Although each container of leachate initially contained at least 2.5 mg/l of Cu, Pb and Hg, no Cu or Hg was detected in any of the samples collected in the lysimeters. Pb was present in steadily decreasing amounts only in samples of raw leachate that were acidified prior to filtration. Apparently, these trace elements were adsorbed onto particulate matter in the leachate or precipi-

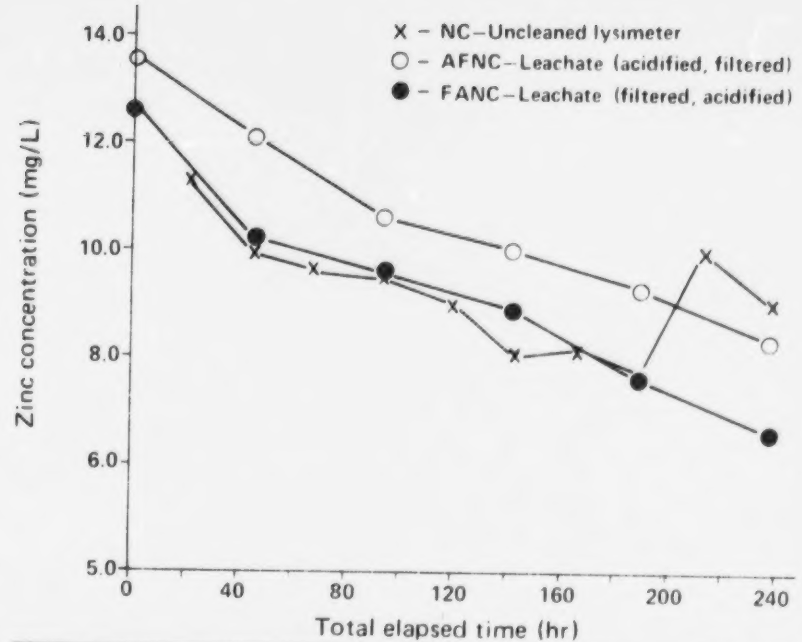


Figure 8. Zinc concentration in samples obtained from an uncleaned lysimeter compared with samples of raw leachate during a 240-hour period.

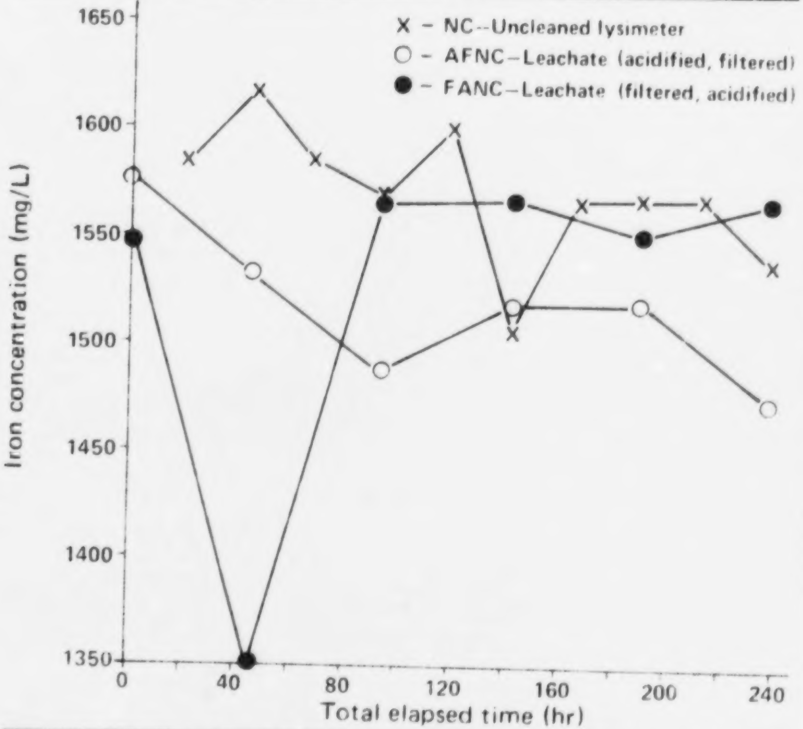


Figure 9. Iron concentration in samples obtained from an uncleaned lysimeter compared with samples of raw leachate during a 240-hour period.

tated and progressively settled out of the suspension. Filtration through a membrane filter or a ceramic cup effectively removed all particles containing adsorbed Pb.

Concentrations of Zn in samples of raw leachate decreased steadily with time (Figure 8). For approximately the first 190 hours, samples from the uncleaned lysimeter (NC) contained levels of Zn very similar to that of the samples of filtered-acidified leachate (FANC). The reason for the change after 190 hours is unknown. Although Zn concentrations appeared to decrease with time and exhibited marked fluctuations, results of statistical tests indicated that there were no signficant differences between the means or the variances of lysimeter samples and raw leachate from the same container. The steady decrease of Zn concentrations in leachate with time probably reflects the progressive settling out of particulate matter containing adsorbed Zn.

Total Fe concentrations were very high and fairly consistent in all samples (Figure 9). The iron was presumed to be virtually all in the ferrous (Fe^{+2}) state owing to the reducing conditions of the leachate. The results indicate that the levels and variation of Fe observed in NC lysimeter samples were in general similar to those of the raw leachate being sampled. Fe concentrations in samples from the cleaned lysimeter were quite variable; however, the mean concentration was nearly identical to that of FAC leachate. The variability of Fe observed in lysimeter samples may have resulted from the use of an air pump for sampling. The introduction of air to the samples may have resulted in partial oxidation and subsequent precipitation of the Fe from solution.

The remaining constituents observed in the leachates included Ca, Mg, Na, K and Cl. The concentrations of each of these constituents were generally very high, representative of leachate only a few years old. The variation of chloride from the NC lysimeter with time is shown in Figure 10. In general, the data indicate a great deal of similarity between the lysimeter and raw leachate samples. Mean concentrations and standard deviations for lysimeter samples closely approximated those of raw leachate samples. Although significant differences were found between the variances of NC and FANC concentrations of Ca, Na, K and Fe, no significant differences were observed between mean concentrations in NC lysimeter samples and FANC leachate samples. Differences in mean concentrations from the acid-cleaned lysimeter (C) and FAC leachate samples were in general not significant.

Although some degree of sorption and screening of ions undoubtedly occurred on the ceramic cup, the effects appeared to be negligible compared to the concentrations of several major ions in the leachate. No consistent, significant differences in concentrations were found between lysimeter and raw leachate samples that could be attributed to sorption or screening by the lysimeter. To reduce the possible effects of the ceramic cups on samples, it is generally recommended that new samples be rinsed first with at least

1 liter of approximately 8N HCl and then rinsed with distilled water (10 to 20 liters). Samples from lysimeters in this experiment were generally very representative of the surrounding leachate composition of several major ions; however, more detailed experiments are required to accurately determine the ability of ceramic-cup samplers to obtain representative concentrations of constituents in contaminated solutions.

Discussion of Waste Disposal in the Unsaturated Zone

The implications of waste disposal and leachate control in unsaturated materials have only recently become of concern to hydrogeologists. Whereas the factors that control saturated ground-water flow are reasonably well known to geologists and engineers, the laws governing unsaturated flow are commonly misunderstood and misapplied. The unsaturated component of the ground-water flow system is neglected in most hydrogeologic studies.

A frequently encountered hypothesis states that leachate will not form or migrate from solid waste emplaced above the zone of saturation. This study and others, however, have shown that infiltration will result in leachate migration through unsaturated materials beneath waste disposal sites in humid climates. Although this zone of partial saturation is generally a good buffer zone, the monitoring and control of leachate can be difficult. Whereas relatively simple techniques, such as wells, commonly can be used to remove and renovate contaminated ground water in saturated materials, such methods are generally not applicable to cases of extensive soil moisture contamination indicative of imminent ground water pollution. Where the underlying unsaturated materials are thick, the migration of the hazardous pollutant usually is detected later than at sites nearer or below the water table and correction can be more involved and costly. If the surrounding soils are sufficiently fine grained to provide adequate renovative capabilities, disposal above the water table would be of limited concern; however, those areas that have a well-defined, deep water table are generally underlain by coarse-grained materials that drain ground water and, potentially, leachate from the site.

Landfill liners and leachate collection systems are also generally designed for use in a saturated system where ground-water flow through fine grained materials is restricted, and water easily flows into a gravel drain from which it can be pumped for treatment. In unsaturated soils, however, infiltration may actually bypass a leachate collection system, especially if the collection system consists of a narrow, coarse-gravel drain overlying a wide liner with a low hydraulic conductivity. In partially saturated soils, water is held under tension in progressively smaller pores; as water contents decrease, tension increases. Thus, the large pores of a gravel drain are the first to be drained with partial dewatering and the last to be filled upon rewetting. Hydraulic conductivity also decreases with a reduction in saturation to the point where it approaches zero in the absence of water-filled pores.

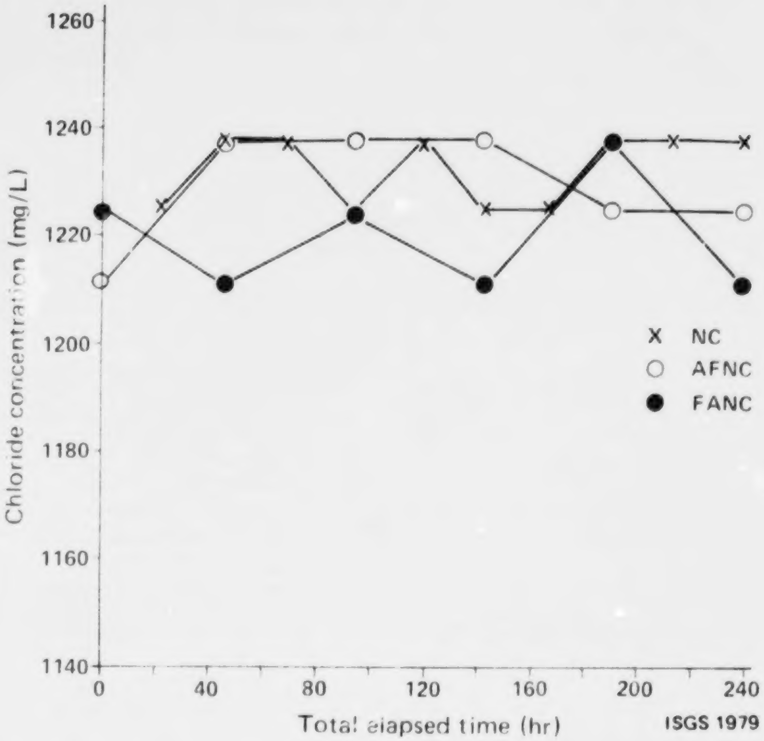


Figure 10 Chloride concentration in samples obtained from an uncleaned lysimeter compared with samples of raw leachate during a 240-hour period

Infiltration through the surface of the landfill will raise the water content of the cover and refuse; however, saturation is not required for downward movement toward the water table. In partially saturated soils, soil moisture may move preferentially through fine-grained materials whose predominantly small pores remain filled with water. An adjacent gravel drain on the other hand, contains a few large pores that require nearly total saturation before soil moisture will enter. In unsaturated materials, therefore, a sump or gravel drain may be dry, but flow may be occurring, and leaching, although at limited rates, may be causing the migration of contaminants from the refuse toward the water table.

Summary

The chemical analyses of samples of soil moisture collected with pressure-vacuum lysimeters indicated the presence of leachate contamination beneath each landfill studied. The majority of the monitoring wells at these landfills have not been affected by leachate. Contamination by leachate was detected almost solely by samples of soil moisture. Given enough time and more careful placement of monitoring wells at these landfills, however, contamination of ground water probably will be observed. The importance of the early detection and delineation of zones of contamination beneath waste disposal sites in unsaturated materials necessitates monitoring of the unsaturated zone.

Soil samples collected for chemical analysis yielded little information regarding leachate migration, and subsequent boring would have been necessary to define changes in soil-water chemistry—an expensive and time-consuming process. The pressure-vacuum lysimeters used in this study have the advantage of providing soil-water samples continuously and repeatedly from unsaturated materials.

There are several limitations in the use of pressure-vacuum lysimeters. The relatively small sample volumes (500 ml or less) limit the selection of parameters for chemical analysis. The ceramic cup may alter the composition of samples and plugging of the ceramic cup pores may reduce the volume of the sample. In addition, sampler failure is, for all practical purposes, irreparable. Preliminary laboratory results indicate that lysimeters may be suitable for collecting representative samples of highly contaminated landfill leachate; however, no detailed experiments on validity of samples have been completed.

Regulations resulting in the increased use of the unsaturated zone for waste disposal should consider the implications of unsaturated flow through and beneath waste disposal sites where standard ground-water monitoring techniques are not applicable. The development of effective procedures to monitor the unsaturated zone will be necessary to implement an adequate monitoring program. Further research is also needed to evaluate the effectiveness of gravel drains and landfill liners as leachate collection systems in unsaturated materials.

Biographical Sketches

***Thomas M. Johnson** is an assistant geologist with the Illinois State Geological Survey in Champaign, Illinois. He earned his M.S. in geology and M.S. in water resource management from the University of Wisconsin-Madison in 1975.*

***Keros Cartwright** is geologist and head of the hydrogeology and geophysics section of the Illinois State Geological Survey in Champaign, Illinois. He earned his Ph.D. in geology from the University of Illinois in 1973.*

***Rudolph M. Schuller** is an assistant geochemist with the Illinois State Geological Survey in Champaign, Illinois. He earned his M.S. in geology from Wright State University in Dayton, Ohio, in 1976.*

Acknowledgments

Funds for this investigation were provided by the Illinois Institute of Natural Resources under Contract No. 40.029. The authors gratefully acknowledge the cooperation and assistance of the landfill operators and the Illinois Environmental Protection Agency.

References

Apgar, M., and D. Langmuir. 1971. Ground-water pollution potential of a landfill above the water table. Ground Water, v. 9, no. 6, pp. 76-96.

Dazzo, F.B., and D.F. Rothwell. 1974. Evaluation of porcelain cup water samplers for bacteriological sampling. Applied Microbiology, v. 27, no. 6, pp. 1172-1174.

Gerhardt, R.A. 1977. Leachate attenuation in the unsaturated zone beneath three sanitary landfills in Wisconsin. Wisconsin Geological and Natural History Survey, Information Circular 35, 93 pp.

Grover, B.L., and R.E. Lamborn. 1970. Preparation of porous ceramic cups to be used for extraction of soil water having low solute concentrations. Soil Science Society of America Proceedings, v. 34, no. 4, pp. 706-708.

Hansen, E., and A. Harris. 1975. Validity of soil-water samples collected with porous ceramic cups. Soil Science Society of America Proceedings, v. 39, pp. 528-536.

Hughes, G.M., R.A. Landon, and R.N. Farvolden. 1971. Hydrogeology of solid waste disposal sites in northeastern Illinois. U.S. Environmental Protection Agency, Solid Waste Management Series, Report SW-12d, 154 pp.

Johnson, T.M., and K. Cartwright. 1980. Monitoring of leachate migration in the unsaturated zone in the vicinity of sanitary landfills. Illinois State Geological Survey Circular 514, 82 pp.

Levin, M.J., and D.R. Jackson. 1977. A comparison of in situ extractors for sampling soil water. Soil Science Society of America Journal, v. 41, pp. 535-536.

Parizek, R., and B. Lane. 1970. Soil-water sampling using pan and deep pressure vacuum lysimeters. Journal of Hydrology, v. IX, no. 1, pp. 1-21.

Wagner, G.H. 1962. Use of porous ceramic cups to sample soil water within the profile. Soil Science, v. 94, pp. 379-386.

Wolfe, R.G. 1967. Weathering of Woodstock Granite near Baltimore, Maryland. American Journal of Science, v. 265, pp. 106-117.

Wood, W.W. 1973. A technique using porous cups for water sampling at any depth in the unsaturated zone. Water Resources Research, v. 9, no. 2, pp. 486-488.

Wood, W. 1978. Use of laboratory data to predict sulfate sorption during artificial ground water recharge. Ground Water, v. 16, no. 1, pp. 22-31.

**Mention of trade names in this paper does not imply endorsement by the Ground Water Monitoring Review, Illinois State Geological Survey or sponsoring agencies.*

A POSITIVE DISPLACEMENT GROUND-WATER SAMPLING DEVICE

By R. W. Gillham and
P. E. Johnson

Introduction

The accurate description of the distribution of a contaminant plume or tracer in geological materials requires a detailed three-dimensional array of sampling points Yare (1975) and Anderson (1977) described procedures that involved the collection of water samples through the center of hollow-stem augers. A major disadvantage of these methods is that no instrument is left in the ground for future monitoring. Alternate procedures, including nests of piezometers and various types of in-line samplers that are installed down the same borehole, are discussed in Pickens et al. (1978).

Pickens et al. (1978, 1981) developed a multi-level sampling device consisting of small screens inserted in the side of a PVC pipe. Each screen was connected to a small-diameter polypropylene tube that passed through the center of the pipe and extended to the ground surface. Samples were collected by applying suction to the ends of the polypropylene tubes. Examples were shown where the devices were used to obtain highly detailed geochemical profiles. The device has three limitations:

- it can only be used in cohesionless geological materials that will cave around the well when it is installed;
- it cannot be used at sites where the water table is below the depth of suction lift (about 8 m);
- it may not be suitable for sampling for dissolved gases or volatile contaminants because the sample is collected under vacuum.

This can lead to uncertainty in parameters that are sensitive to the dissolved gas composition such as pH, alkalinity, redox potential and redox-dependent trace metals.

Cherry et al. (1980) described a bundle-type piezometer that consisted of several 0.95 cm I.D. polyethylene tubes arranged around a central 1.27 cm I.D. schedule 80 PVC pipe. The tubes were terminated at different depths and each tube was fitted with a short screened section at the bottom. The bundle was installed through the center of hollow-stem augers. Because of the larger diameter of the tubes, water samples can be collected from depths greater than the limit of suction lift by using a small-diameter tube as a bailer or by modifications of the pneumatic sampling method described by Sommerfeldt and Campbell (1975). In both sampling methods, contact between the sample and the atmosphere will occur, giving rise to uncertainty regarding the volatile constituents of the sample.

This paper describes a modification of the Pickens et al. (1978, 1981) device. The modification allows water to be collected through the use of positive pressure rather than suction. As a result, samples can be collected in situations where the water table is at a substantial depth and the potential for loss of volatile constituents of the ground water is greatly reduced.

Description of the Sampling Device

The sampling device is similar to that described by Pickens et al. (1978, 1981) in that it consists of a PVC pipe with small screened areas inserted at vertical intervals along the pipe. It differs, however, in that each intake screen is fitted with a pumping device that is operated by air pressure. Figure 1 shows a schematic of a typical installation. Figure 2 shows details of an individual pumping device.

The screen assembly is identical to one of the types described by Pickens et al. (1978) and consists of a stainless steel screen disc heat-welded to the end of a 2.0 cm length of 1.9 cm I.D. schedule 80 PVC pipe. The screened intake pipe is glued into the wall of a 5.0 cm I.D. PVC pipe. To prevent the movement of fine materials through the screen and into the check-valve assembly, a layer of glass-fiber wool is placed over the sampler screen. A No. 00 rubber stopper (cut in half) is used to attach the sampling point to a length of 0.32 cm O.D. polyethylene tubing. This tubing is attached to a plastic check-valve designated as A in Figure 2 that is joined to a polyethylene "T" connector. One outlet of the "T" is connected to a second check-valve designated as B in Figure 2 while the second outlet is connected to a 50 cm^3 plastic syringe. The outlet of check-valve B is connected to a sampling tube leading to ground surface.

The syringe plunger was modified by cutting off the plastic handle just above the plunger. A No. 6 rubber stopper containing a short length of stainless steel tubing was fitted securely into the top of the plunger. A brass retaining plate passed over the steel tubing and was fastened to the finger grips of the syringe with machine screws. The finger grips were trimmed to the edge of the retaining plate to reduce the diameter of the syringe. The stainless-steel tube emerging from the top of the syringe was connected to the 0.3 cm polyethylene pressure tube leading to ground surface. All tubing connections that would be subject to the applied positive pressure were made using Gyrolok tube connectors. In order to accommodate the syringes and tubing connections, the minimum distance between adjacent sampling points is approximately 30 cm.

To operate the sampler, a source of both negative and positive air pressure is required. Upon the application of negative pressure, the syringe plunger rises causing valve B to close, valve A to open and water to be drawn through the screen and into the syringe. With the application of positive pressure the plunger is forced down, closing valve A, opening valve B and forcing the sample up the sampling tube to ground surface. The operation can then be repeated until the required sample volume has been collected.

The sampler described above was constructed from inexpensive and readily available components, without any serious attempt to design for maximum depths of operation. Nevertheless, laboratory tests showed the device to successfully sustain pressures of five atmospheres (higher pressures were not attempted). This is the approximate pressure that would be required to operate a syringe located at a depth of 50 m below ground surface. Thus the design used in developing the prototype should be adequate for the majority of ground-water contamination problems.

If the water table is less than about eight to 10 m below ground surface, a positive-negative hand pump can be used to provide the necessary gas pressures. If at greater depths, it is more convenient to use a small compressed-air cylinder and regulator to provide the positive pressure. Only small negative pressures are required; and generally, if the syringe is located at a depth of three to four m or more below the water table, it will fill without the application of negative pressure as a result of the hydrostatic pressure in the aquifer. By connecting all pressure lines to a single manifold and the sampling lines to individual sample containers, the sample points of an individual well can be sampled simultaneously.

The devices constructed to this point have been experimental in nature. It is therefore difficult to arrive at the real cost of an installation. The components for an individual pumping device and sampling point cost approximately $10; and if an installation has a total depth of 25 m, with 10 sampling points spaced over the bottom 10 m, the cost of the PVC pipe and polyethylene tubing would be on the order of $225. Thus the cost of materials would be about $325. It is estimated that the time required for an experienced technician to completely assemble the unit would be 20 hours. Depending upon the cost per hour for labor, construction of the entire sampling well would cost on the order of $500 to $600. Although the cost is significant, it is small compared with the drilling, labor and materials cost for 10 individual piezometers installed over the same depth range.

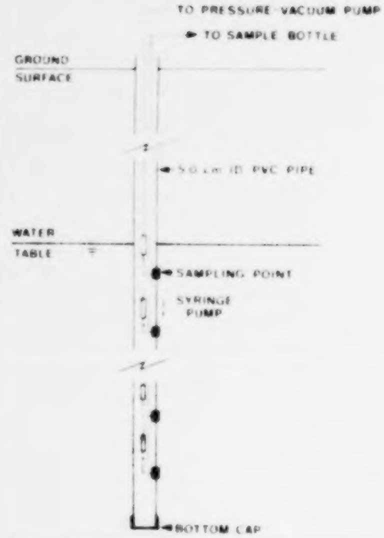


Figure 1 Typical installation schematic

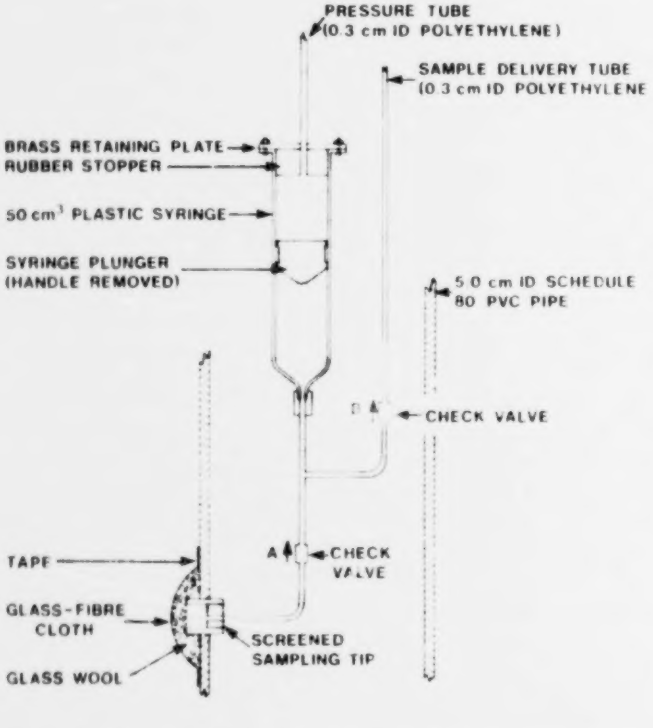


Figure 2 Individual pumping device

Operational Experience

The first positive displacement device was installed through an abandoned sanitary landfill site on the Canadian Forces Base at Borden, Ontario, in July 1978. Installation was through the center of 8.3 cm I.D. hollow-stem augers. At the time, the water table was about eight m below ground surface, or about at the limit of suction lift. Eight sampling points were used and all performed normally. With periodic sampling, only four of the sampling points were operating by the fall of 1979 and the device was lost to vandalism in the spring of 1980.

Two additional devices installed in the summer of 1979 gave less satisfactory results. Initially only half the points yielded water, and by the fall of 1979 only one or two points on each well were operating.

Once a point ceases to operate, the pumping device cannot be retrieved for repair and thus the point is effectively lost from the sampling network. Thus, the operational results to the fall of 1979 were not encouraging. However, the design used in the initial installations differed substantially from that shown in Figure 2. The method of connecting the pressure line to the syringe is improved, and improved methods for making the tubing connections are being used. In addition, the initial installations used a ball-type of check-valve rather than the pinch-type used in the present design. Laboratory tests showed that very small amounts of sediment in the sample would prevent the ball from seating and thus prevent the device from operating. The pinch-type valves are less susceptible to this problem. The original design also did not have the glass wool over the intake screen. Because of the arrangement of check-valves, there is no opportunity to flush the screen should it become clogged. The glass wool and glass-fiber cloth should act as an additional filter. The glass wool provides another advantage by providing a larger surface area; thus the velocity of the water in the formation is reduced and there is a reduced tendency for the fine materials to migrate.

Three installations were made in the summer of 1980 using the design of Figure 2. One was at the Woolwich landfill located about 15 km north of Waterloo, Ontario. The geological material at this site was medium sand and the water table was located at a depth of 20 m below ground surface. The total depth of the installation was 28 m. The other two devices were installed in sandy materials near a waste lagoon located about 12 km southwest of Waterloo. The depth to the water table was about 15 m. Periodic sampling has shown all the points of the three new devices to operate normally.

The positive-displacement sampler installed in 1980 at the Woolwich landfill site was located adjacent to a bundle-piezometer of the type described in Cherry et al. (1980). The positive-displacement sampler has proven to be much easier and faster to sample.

There is no opportunity for measuring hydraulic head using the sampling tube of the positive displacement sampler. In one of the installations of 1980, additional screens were located in the wall of the pipe and connected directly to 1.27-cm I.D. polyethylene tubing that was terminated at ground surface. Water-level measurements can be made in these tubes using a small-diameter water level contact gauge.

Discussion

The positive-displacement sampler provides a fast, convenient and economical means of obtaining "point" water samples from different depths in the same well. Its operation is not limited by the depth to the water table, thus providing a considerable advantage over the Pickens et al. (1978, 1981) device. In addition, the sample remains under positive pressure and is not exposed to the atmosphere until it reaches the surface. By connecting the sample line to a suitable sample container, such as a syringe, the sample can be entirely isolated from the atmosphere. The device should therefore be an effective means of sampling for volatile ground-water constituents and dissolved gases. For problems of this nature, it may be the most appropriate device to use even in situations with shallow water tables.

It should be noted that the use of the positive displacement sampler, as well as devices of a similar type, is restricted to geological materials that will cave around the well and that will yield water to the sampling point at a reasonable rate. Because of the check-valve arrangement, the positive-displacement sampler will undoubtedly be more susceptible to problems caused by silt migration through the screen than the other samplers referred to in this article. Our experience to this point has been entirely in relatively clean, medium to fine sand geological materials.

The success of the 1980 installation is qualified only by the short period over which they have been sampled. We anticipate that additional installations will be made at the Woolwich landfill site in 1981 as part of the routine monitoring network.

Acknowledgements

Dr. John Cherry of the Department of Earth Sciences at the University of Waterloo provided many helpful suggestions during the development of the instrument described in this paper and during preparation of the manuscript. Funds for the work were provided through the Canadian Department of Supply and Services (contract No. OSU 78-00195) and the National Science and Engineering Research Council. The plastic check-valves used were "Harleco" Flo-Check valves manufactured by Harleco, A Division of EM Industries Inc. The plastic syringe used was a Plastipak Disposable Syringe, 50 cc with Luer-Lok tip.

References

Anderson, E.G., 1977. New ground-water contaminant sampler for contaminant plume mapping. Canadian Water Well, p. 36.

Cherry, J.A., Gillham, R.W., Anderson, G.E. and Johnson, P.E., 1980. CFB Borden landfill study Volume 1: Hydrogeological studies of a sandy aquifer at an abandoned landfill. Part 2: Ground-water monitoring devices. Final report submitted to the Canadian Department of Supply and Services under Contract No. OSU-78-00195.

Pickens, J.F., Cherry, J.A., Grisak, G.E., Merritt, W.F. and Risto, B.A., 1978. A multi-level device for ground-water sampling and piezometer monitoring. *Ground Water*, V. 16, No. 5, pp. 322-327.

1981. A multi-level device for ground-water sampling. *Ground Water Monitoring Review*, V. 1, No. 1, pp. 48-51.

Yare, B.S., 1975. The use of a specialized drilling and ground-water sampling technique for delineation of hexavalent chromium contamination in an unconfined aquifer, southern New Jersey Coastal Plain. *Ground Water*, V. 13, No. 2, pp. 151-154.

Sommerfeldt, T.G. and Campbell, D.E., 1975. A pneumatic system to pump water from piezometers. *Ground Water*, V. 13, No. 3, pp. 293.

Biographical Sketches

Dr. Robert W. Gillham is an associate professor in the Department of Earth Sciences at the University of Waterloo. His principal area of interest concerns the migration of contaminants through geological materials.

Paul E. Johnson is a research technician in the Department of Earth Sciences at the University of Waterloo.

RECOMMENDED SAMPLING PROCEDURES FOR MONITORING WELLS

By Rudolph M. Schuller, James P. Gibb and Robert A Griffin

Introduction

Monitoring wells are the principal source of data for evaluating the effects of waste disposal sites on ground water. For these evaluations to be meaningful, the water samples collected from the monitoring wells must be representative of the water in the aquifers or water-bearing strata being studied. Although several laboratories and agencies have proposed procedures for sampling water from monitoring wells, there exists little supportive data to indicate that these procedures produce "representative" water samples.

The objective of this study was to develop a sampling protocol for monitoring wells. Included in this protocol are:

(1) criteria for selection of an appropriate pumping mechanism;
(2) procedures for determining the necessary extent of well flushing before a sample should be collected;
(3) procedures for sample preparation, preservation and storage.

The following is a summary of the pertinent findings of this study. The complete documentation of the results is available in the final report to the U.S. Environmental Protection Agency, Cincinnati, Ohio.

Experimental Design

Six sites in Illinois, representing a range of hydrologic conditions and chemical characteristics of aquifer water, were chosen for study. Two of the sites are currently operating sanitary landfills (sites 3 and 4); two are inactive sanitary landfills (sites 2 and 6); one is a stack scrubber lagoon for a secondary zinc smelter (site 1) and one is an anaerobic digestion lagoon for a hog processing plant (site 5). Five of the wells sampled are cased with PVC pipe, either 3.8 or 5.1 cm (1½ or 2 in.) in diameter. The sixth well is cased with 5.1 cm (2 in.) diameter galvanized iron. The wells have depths ranging from 5 to 10 m (16 to 30 ft.) and have nonpumping water levels from 0 to 5 m (0 to 16 ft.) below land surface.

Samples were obtained from the monitoring wells by four methods. These included a peristaltic pump, an air-lift system, a nitrogen-lift system and a bailer. The mechanical pump was a Master-flex 7545 variable-speed drive unit equipped with a model 7015 peristaltic pump head. When operated on 115 volts (a portable generator was used in the field), the pump is capable of producing from about 50 to 1,000 mL/min. For air-lift and nitrogen-lift pumping systems, an apparatus similar to that shown in Figure 1 was used. For some sampling intervals, a 1.27 cm (½ in.) diameter rigid PVC discharge pipe and 0.635 cm (¼ in.) plastic air line were employed. For others, a 0.952 cm (⅜ in.) diameter flexible polyvinyl discharge line and 0.635 cm (¼ in.) plastic air line were used. For all air-lift pumping, a four-cylinder, electric-driven air compressor was used, and for the nitrogen-lift pumping, a cylinder of compressed nitrogen was used. A pressure regulator and flowmeter controlled the pressure and flow rate to the gas line. A 2.54 cm (1 in.) diameter stainless steel bailer, 0.914 m (3 ft.) long, which retrieved a sample of about 350 mL, was constructed to bail the test wells.

While collecting samples with the peristaltic, air and nitrogen-lift systems, the pump intake was placed in the middle of the screened portion of the well. The bailer was lowered to the same depth as the pump intakes for the other systems.

Prior to the collection of samples, controlled pumping tests were conducted in an attempt to determine the hydraulic characteristics of the materials in which the wells were finished. An understanding of the yield potential of the well and the transmissivity of the "aquifer" being sampled is essential to devising a proper sampling scheme. Without knowledge of these factors, decisions cannot be made concerning the pumping rate or the length of time the well should be pumped prior to collecting a "representative" sample.

Traditional analyses of pump test data use the equations derived by Theis (1935) and Jacob (1950). One of the basic assumptions made in those equations is that all of the water pumped from a well during the pumping test comes from the aquifer and none from storage with-

in the well. Because this condition is not always fulfilled in practice, particularly for low-yielding wells commonly used for monitoring, the Theis and Jacob equations are inappropriate for describing the behavior of water levels during pumping for most shallow, low-volume monitoring wells.

Papadopulos and Cooper (1967) presented an equation describing the discharge from a pumped well that takes into account the volume of water removed from casing storage. Drawdown values calculated from this equation differ significantly from those from the Theis and Jacob equations during the early portion of the pumping test when a relatively high percentage of the discharge comes from casing storage. During the later stages of the pumping test, when only a negligible quantity of water is obtained from casing storage, the equations produce equivalent results.

Data from the pump test run on the well at site 6 are given here as an example of the results obtained. The well was pumped with a peristaltic pump at rates from 765 to 620 mL/min. for a period of two hours. Recovery measurements were taken for 10 minutes after pumping stopped. With the Papadopulos and Cooper method of analysis, an aquifer transmissivity of about 2.48m^2/day (200 gpd/ft.) was determined.

Once the transmissivity of the aquifer tapped by a monitoring well has been determined, the percentage of water coming from the aquifer and that from storage can be determined as a function of pumping time. Figure 2 illustrates the percentage of water pumped that is coming from aquifers of different transmissivities at a pumping rate of 500 mL/min. As expected, the higher percentages of aquifer water are derived from the higher yielding wells (larger T values). Once a T value of 62.0 m^2/day (5,000 gpd/ft.) is reached, larger values of T make little or no difference. Thus, if the yield capability of the well is sufficiently large (T > 62.0 m^2/day), the **effects of water from storage become negligible for 5.08 cm (2 in.) diameter wells and can be ignored when establishing a sampling protocol.**

Additional calculations illustrate the effect of the diameter of the well on the percentage of aquifer water removed at a constant pumping rate and transmissivity (Figure 3). For a T of 2.48 m^2/day (200 gpd/ft.), well diameters larger than 5.08 cm (2 in.) have a significant effect on the percentage of aquifer water pumped, particularly during the very early stages of pumping. Results from analyses of pump test data provide the basic hydrologic information needed to permit a better understanding of the changes in the chemical quality of the water, as a function of pumping time, that are likely to occur as the sample is pumped from a monitoring well.

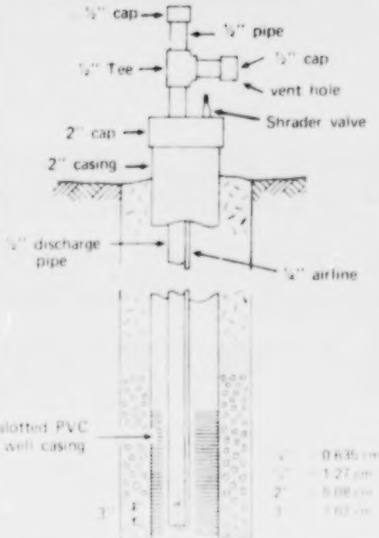


Figure 1. Typical well and air-lift pumping mechanism.

Each of the six sites was initially sampled a maximum of once a month by one of the four pumping mechanisms. The frequency of sampling allowed the wells to recover from pumping during the previous sample collection and simulated the sampling routine recommended by many regulatory agencies.

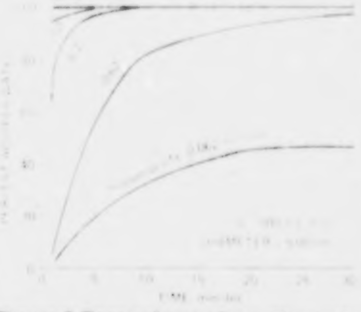


Figure 2. Percentage of aquifer water versus time for different transmissivities.

Samples were taken from the initial water in storage, designated as the zero well volume and at ½, 1, 1½, 2, 4, 6, 8 and 10 well-volume intervals. One well-volume is defined as the amount of water occupying the well casing before pumping is initiated. Measurements of pH, specific conductance and alkalinity were made on unfiltered aliquots of each of the well volumes or partial well volumes that were sampled. These measurements were conducted immediately after the aliquots were collected to minimize opportunities for changes in chemical composition due to exposure to atmospheric conditions.

The sample aliquots were subdivided into three portions by filtering through either a 3.0 μm, 0.45 μm or 0.22 μm pore-size membrane. Plastic filter holders fitted for 47 mm diameter filters were used with compressed nitrogen gas overpressure to speed filtration. All filtering and preservation was conducted in the field immediately after each sample was pumped to the surface. Procedures used for sample preservation are those listed in the U.S. Environmental Protection Agency's "Methods for Chemical Analysis of Water and Wastes" (1979). All samples were stored in linear polyethylene bottles. The samples were analyzed for more than 20 constituents by inductively coupled, argon-plasma-source, direct-reading spectrometry (ICP), ion-specific electrodes and colorimetric techniques. Only the results of the major constituent analysis are reported in this summary of the study.

An additional sampling scheme was conducted to further verify the effects of sample-storage time on sample integrity. At two of the sites, 4-L volume samples were collected

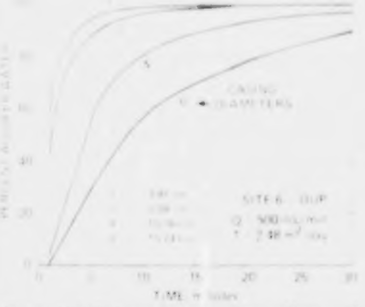


Figure 3. Percentage of aquifer water versus time for different well casing diameters.

from the 0, 1, 2, 4, 6, 8 and 10 well-volumes pumped with a peristaltic pump. A small aliquot of each of the collected samples was immediately filtered and preserved. After pumping 10 well-volumes, the pump was withdrawn from the well and replaced with the air-lift mechanism and a second 4-L sample was taken by using compressed air. Nitrogen gas was then substituted for the air and was used to collect another 4-L sample. The air-nitrogen-lift mechanism was then removed from the well and a 4-L sample was collected using a bailer. On-site pH measurements were made on each of the unfiltered samples. The remainder of the 4-L samples were returned to the laboratory unfiltered and without preservation; additional aliquots were taken the same day as field sampling and every succeeding day for a period of four days.

Table 1. Analysis of Samples Collected Sequentially by Four Pumping Mechanisms Immediately After Flushing 10 Well-Volumes (Concentration in mg/L)

Site 5: Anaerobic lagoon

Sequence	Pumping Mechanism	pH	Ca	Fe	K	Mg	Na	Zn
1	Peristaltic	6.7	111	11.6	ND*	44.6	186	0.18
2	Nitrogen-lift	7.7	106	3.7	ND*	44.0	183	0.03
3	Air-lift	7.5	105	0.7	ND*	43.6	182	0.03
4	Bailer	6.8	105	13.5	ND*	42.2	179	0.96

Site 6: Inactive sanitary landfill

Sequence	Pumping Mechanism	pH	Ca	Fe	K	Mg	Na	Zn
1	Peristaltic	6.8	32.5	5.74	189	94.2	215	ND*
2	Nitrogen-lift	7.8	36.1	1.79	202	111	239	ND*
3	Air-lift	7.8	37.1	.57	199	115	247	ND*
4	Bailer	6.8	42.3	7.42	193	114	235	ND*

*ND—Not detectable

Results

Pumping Mechanisms

During the monthly sampling routine used in this study, the samples collected with the peristaltic pump and bailer were discovered to have pH values 0.5 to 1.0 pH units lower than those samples collected with the nitrogen- and air-lift systems. The sampling routine using all four pump mechanisms was then conducted to verify that the change in pH was due to the pump used and not seasonal variations over the initial four-month sampling period.

The pH values for the samples collected by all four mechanisms at two of the sites are given in Table 1. The pump systems are listed in the order that they were used at the sites. Again, the pH values of the samples taken with the peristaltic pump and bailer are as much as 1.0 pH unit less than those samples taken with the air- and nitrogen-lift systems. The increased pH of samples collected with the nitrogen and air-lift systems is a result of bubbles of air and nitrogen rising through the water in the discharge pipe and stripping dissolved CO_2 from the water. The loss of CO_2 results in an increase in pH.

Table 1 also shows that the Fe and Zn concentrations at both sites were affected by the choice of pumping system used to collect the samples. The pump system used does not appear to affect the concentration of Ca, K, Mg, Mn or Na. The loss of Fe from solution is most probably due to the precipitation of hydrous Fe oxides. Adsorption of Zn onto the Fe precipitates is presumed to be responsible for the loss of Zn from solution. At both sites the dissolved Fe concentrations found in the bailed samples were higher than in the samples taken with the peristaltic pump. This is believed to be a result of precipitated Fe from the air- and nitrogen-lift pumping sequences being taken back into solution by the lower pH water entering the well from the aquifer and subsequently collected with the bailer.

For all sites the highest Fe and Zn concentrations were found in samples that had been collected with either the peristaltic pump or the bailer. The choice of pump systems had little or no effect of the Ca, Cl^-, F^-, K, Mg, Mn or Na concentrations. The results of this phase of the study indicated that the peristaltic pump produced the most reproducible and representative data from the monitoring wells of the four pump systems used. The use of a bailer is acceptable as long as precautions are taken to regulate the rate and duration of bailing and to prevent contamination.

Well Flushing

It is a common practice to flush a monitoring well to remove the stagnant water (storage water) in the well casing before sample collection. Although this is a common practice, there is again little supportive data to establish the extent of well flushing necessary before a representative sample can be taken from a particular monitoring well.

Figure 4 is a plot of the Fe, K, Mg, Mn and Zn concentrations in samples collected with a peristaltic pump from site 5 and is representative of the results obtained during this phase of the project. Figure 4 shows that there is no significant change in the K, Mg or Mn concentrations with volumes pumped. However, the Fe and Zn concentrations in the water decreased steadily until 8 to 10 well-volumes had been flushed from the well. Overall, for the six sites studied, the concentrations of Ca, Cl^-, K, Mg, Mn and Na showed little or no change as a response to well flushing. The extent of flushing did affect the concentrations of Fe and Zn at all of the sites.

The individual worker must recognize the existence of site-specific variables of geology, hydrology and water chemistry between waste disposal sites, and realize that the extent of well flushing required to obtain a "representative" water sample will be different for each monitoring well. A specific volume of water pumped from one monitoring well before sample collection may be sufficient to produce a representative sample of the aquifer water, but at another site the same volume may be insufficient or result in overpumping. Overpumping may produce ground water from another source that could dilute or concentrate certain constituents. This is especially a problem at wells in disposal sites where heterogeneous wastes are buried in close proximity to the well. There may also be site-specific variations in the response of a particular constituent to well flushing.

Seasonal Fluctuations in Quality of Ground Water

A problem that may arise during sampling involves seasonal fluctuations in the quality of ground water. Two of the sites were sampled twice using the peristaltic pump, once during the monthly sampling and a second time when all four pump systems were used sequentially. It is instructive to compare the samples that were collected for one of these sites using the same pump under identical operating conditions but at different times of the year.

Figure 5 depicts the change in Fe concentrations while pumping 10 well-volumes at site 6 during the August and February sampling intervals. The well at site 6 is screened directly beneath the refuse at a municipal landfill. The landfill has a thin soil cover, which allows for rapid recharge by infiltrating rain water. These factors increase the possibility of seasonal variations in ground-water chemistry at the site. Ground water within a landfill also has an extremely heterogeneous chemical nature, dependent upon its immediate environment within the landfill. Thus, samples from a well placed directly into or beneath the refuse could exhibit large fluctuations in chemical composition.

The decrease in Fe during pumping in August was felt to represent an aquifer under oxidizing conditions. The trend of increasing Fe in the February samples would indicate that the aquifer (or refuse) was under a more reduced environment with more Fe in solution. The presence of a frozen soil cover during winter, creating a closed system and not permitting the introduction of air (oxidants), could explain the reducing environment. It appears that it would be very difficult to obtain reproducible "representative" samples of water from this monitoring well, as there are too many factors affecting the quality of the water.

Sample Preparation and Storage

The effort to collect a representative ground-water sample from an aquifer via a monitoring well would be futile if the chemical composition of the sample changed between the time of collection and analysis. Proper sample preparation, preservation and storage can help prevent such changes from occurring. This phase of the project was primarily concerned with whether the U.S. EPA-prescribed procedures (1979) for preparation and preservation of samples should be applied on-site immediately after sample collection or later in the laboratory.

Table 2 gives the results of the storage analysis conducted on samples taken from sites 5 and 6 respectively. Again, certain parameters are more sensitive than others to the length of time before filtration and preservation. The sample pH increases 0.3 to 0.4 units, and the Fe and Zn concentrations are all but reduced to less than detectable concentrations in a 7-hour period between collection and preparation. The Ca, K, Mg, Mn and Na concentrations display a slight decreasing trend over a 48-hour period.

Results of the filtration experiments using the three pore-sized membranes are not included in this review. However, the analyses indicated that Fe, Mn and Zn were sensitive to the pore size of the membrane used for filtration and Ca, K, Mg and Na concentrations were similar regardless of the pore size. The 0.45 μm pore-sized membrane effectively removed clay and colloidal-sized particles and still resulted in a practical filtration rate for on-site use.

Conclusions

On the basis of the results of this study, collecting "representative" water samples from monitoring wells is not a straightforward or easily accomplished task. Each monitoring well has its own individual hydrologic and chemical character that must be considered when planning a sampling protocol. The selection of the type of sampling device, the sample preparation, preservation, storage and the sampling procedures must all be tailored to the size and accessibility of the individual well, its hydrological and chemical character, the chemical constituents of interest, the time of year and the purpose for monitoring.

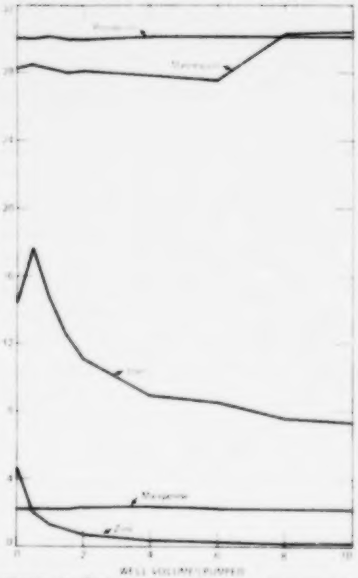


Figure 4. Iron, potassium, magnesium, manganese and zinc concentrations versus volumes pumped with a peristaltic pump (site 5).

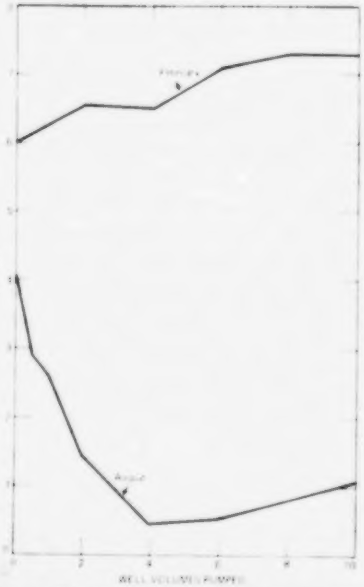


Figure 5. Iron concentrations versus well-volumes pumped for two sampling periods using a peristaltic pump at site 6.

Table 2. Analysis of the 10th Well Volume Sample Used for Storage Study (Concentration in mg/L)

Site 5: Anaerobic lagoon

Hours after collection—before preservation	pH	Ca	Fe	K	Mg	Mn	Na	Zn
0	6.7	111	11.6	32	44.6	3.81	186	0.18
7	7.0	110	.33	31	41.9	3.15	172	0.02
24	7.0	104	< .03	35	41.4	3.10	181	0.02
48	7.0	99	.03	33	39.5	2.98	171	0.02

Site 6: Inactive sanitary landfill

Hours after collection—before preservation	pH	Ca	Fe	K	Mg	Mn	Na	Zn
0	6.8	32.5	5.74	189	94.2	ND*	215	ND*
7	7.2	35.2	< .08	203	110	ND*	242	ND*
24	7.2	32.0	< .08	204	96.6	ND*	223	ND*
48	7.2	30.1	.34	184	89.1	ND*	204	ND*

*ND—Not detectable

The following is a list of general recommendations for sampling procedures that will allow for the collection of representative water samples from monitoring wells when applied on a site-by-site basis.

(1) A brief, two or three hour pumping test should be conducted on each monitoring well to be sampled. Analyses of the pump-test data and other hydrologic information can be used to determine the frequency at which samples will be collected and the rate and period of time each well should be pumped prior to collecting the sample.

(2) Pumping 4 to 6 well-volumes in most cases will produce samples representative of aquifer water. However, calculations of percent aquifer water with time provide a more rational basis on which the length of pumping can be determined. The rate and time of pumping should be determined on the basis of the transmissivity of the aquifer and the well diameter. Samples should be collected in the minimum time required to produce "representative" aquifer water.

(3) Based on the sensitivity of the selected chemical "indicators," a choice of pumps for routine sampling can be made. The use of air- or nitrogen-lift pumping mechanisms is not recommended and, if used, should be restricted to pH-insensitive and nonvolatile chemical constituents. Although this study dealt with inorganic constituents, the data suggests that these types of pumping mechanisms probably would also strip volatile organic compounds from the water during pumping. The peristaltic or submersible diaphragm pumps are recommended for most applications. The bailer would be a second choice.

(4) Measurements of pH, Eh and specific conductance should be made at the time of sample collection. The measurements should be made in a cell closed to atmospheric conditions. All samples should be promptly filtered through a 0.45 μm pore-size membrane and preserved according to recommended U.S. EPA procedures for the chemical constituents of interest.

Acknowledgements

The authors wish to acknowledge partial support under Grant No. R806304 from the U.S. Environmental Protection Agency, Cincinnati, Ohio; Don Sanning, project officer. Mention of trade names does not constitute endorsement by the respective agencies.

The authors also wish to thank William Roy, Ivan Krapac, William Bodner and Michael O'Hearn for their help in various aspects of this project.

References

Jacob, C. E. 1950. Flow of ground water. Engineering Hydraulics. Edited by H. Rouse. John Wiley and Sons, New York.

Papadopulos, I. S. and H. Cooper. 1967. Drawdown in a well of large diameter. Water Resources Research. v. 8, no. 2, pp. 241-244.

Theis, C. U. 1935. The relation between the lowering of the piezometric surface and the rate and duration of discharge of a well using ground-water storage. TransAmerican Geophysical Union. v. 16, pp. 518-524.

U.S. Environmental Protection Agency. 1979. Methods for chemical analysis of water and wastes. EPA-600/4-79-020, Cincinnati, Ohio.

Biographical Sketches

***Rudolph M. Schuller** is an assistant geochemist with the Illinois State Geological Survey in Champaign, Ill. He earned his M.S. in geology from Wright State University in Dayton, Ohio, in 1976.*

***James P. Gibb** is head of the Ground-Water Section of the Illinois State Water Survey in Champaign, Ill. He received his B.S. in civil engineering from the University of Illinois in 1966.*

***Robert A. Griffin** is a geochemist and head of the Geochemistry Section of the Illinois State Geological Survey in Champaign, Ill. He earned his Ph.D. in soil chemistry from Utah State University in Logan, Utah, in 1973.*

Dup

DEVISING A GROUND-WATER MONITORING STRATEGY FOR A GEOLOGIC REPOSITORY FOR RADIOACTIVE WASTE

Leo S. Leonhart, Frank A. DeLuca, N. Thomas Sheahan, Lawrence M. West

The ever-growing stockpile of nuclear waste materials has urged the nation to explore technologies appropriate for the safe isolation of such materials from the human environment. Thus the National Waste Terminal Storage (NWTS) Program of the U.S. Department of Energy (DOE) was chartered to investigate feasible alternatives relating to radioactive waste disposal. Among the alternatives currently being considered by the NWTS Program is that of terminal isolation within repositories mined in deep, stable geologic formations. The properties of various geologic media are presently under study as candidate hosts for such a facility. One of the principal geologic media being studied is that of flood basalt, which is areally extensive in the Columbia Plateau region of the northwestern United States (Figure 1).

The NWTS-sponsored project involved with the study of the Columbia River basalt is called the Basalt Waste Isolation Project (BWIP). BWIP began in 1976 and is currently operated for the DOE by Rockwell Hanford Operations, a division of Rockwell International. For the past five years, the BWIP has carried out various scientific and engineering field studies which have provided significant technical inroads toward understanding the local and regional geologic and hydrologic systems present within the Columbia Plateau and, more specifically, the Pasco Basin (Figure 2). These and other studies have resulted in the selection of a "reference repository location" (RRL) within the Hanford Site. The Hanford Site is situated within the Pasco Basin and represents a nearly 1,500-km^2 (570-mi^2) facility owned and operated by the DOE. Since 1943, this land has been dedicated to research in the use of atomic energy. Areally, the RRL occupies about 45 km^2 (17 mi^2) and is presently the focus of various site characterization studies including the planning of a baseline ground-water monitoring program.

This paper represents a topical treatment of the subject of ground-water monitoring as it relates to the particular needs of high level nuclear waste disposal facilities using the BWIP as a specific reference. While the involvement with management of high-level radioactive wastes and the design and operation of repository facilities is presently parochial to the federal government and certain prime contractors, it is believed that the technical aspects involved with this ground-water monitoring example provide an interesting comparison with those encountered at near-surface and underground-injection, hazardous waste disposal operations. In particular, the integration of several program facets ranging from baselining parameters to validation of predictive models into a comprehensive strategy may be of interest.

Ground-Water Monitoring Requirements

The concept of a geologic repository involves a system incorporating multiple barriers (both engineered and as part of the natural system) designed to achieve safe isolation of the wastes from the human environment throughout the hazardous life of the wastes.

Facility operators must devise a means of evaluating and validating whether the system will meet its performance goal. With regard to hydrologic performance, ground-water monitoring is a potentially important means of achieving the required evaluation/validation. The BWIP has thus embarked on an effort to identify considerations which may be relevant to devising a ground-water monitoring strategy. It is the goal of this effort to develop a conceptual strategy based on feasible alternatives for meeting various objectives which were determined in consideration of both regulatory requirements and the technical and practical realities encountered. Aspects associated with each of these items as they relate to the BWIP are discussed here.

Regulatory Criteria

Recognizing the importance of a repository system performance validation program, the U.S. Nuclear Regulatory Commission (NRC), which will be the lead licensing authority for such repositories, has required in its proposed rulemaking (10 CFR 60 *Federal Register*, Vol 46, No. 130, July 8, 1981, p. 35280-35296) that a performance validation program be developed to ensure that:

- **Actual subsurface conditions encountered and changes in those conditions which occur during construction and waste emplacement operations are compared with conditions assumed during licensing review.**
- Engineered systems and components required for repository operation or which are part of the permanent barrier system are functioning as intended and anticipated.

Because the concept of a subsurface repository involves siting the facility within the "zone of saturation," ground-water monitoring clearly is a key item of the performance validation strategy envisioned by the NRC.

In reading the NRC's proposed drafts of 10 CFR 60 and various attendant guidelines, certain key criteria useful to ground-water monitoring strategy development became evident. Examples of such criteria items include:

- Timing—It is intended that such programs begin during the site characterization stage of the project and continue throughout the licensing period.
- Constraints—The monitoring system should not impair the functioning of natural or engineered systems which are components to contaminant migration barriers.
- Purpose—It is clear that there are multiple purposes intended with respect to monitoring and they include:
  - To establish a reference baseline.
  - To evaluate project impacts.
  - To calibrate and validate models.
  - To provide optional safeguards.

Additionally, public reassurance is an implied purpose.

- Parameters—At least with respect to the establishment of a reference baseline, it is clear that both potentiometric and hydrochemical data are to be monitored sufficiently to ascertain seasonal and long-term variations and that these types of information will be required for each principal hydrogeologic unit.

Because these regulatory criteria emanate from a generic context, intended to provide "umbrella" guidelines for a wide variety of natural conditions and engineered designs, it becomes necessary to transpose the criteria into a syntax specific to the technical aspects of the BWIP.

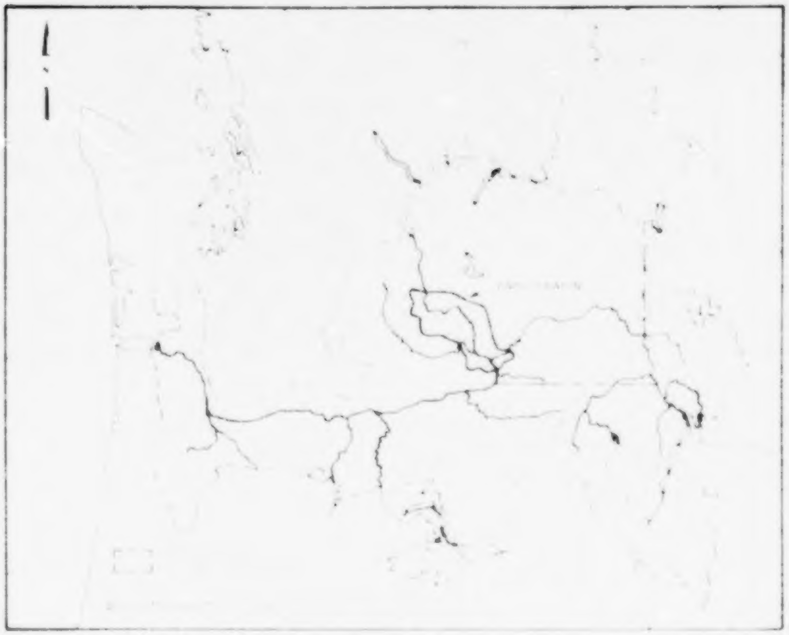


Figure 1 Areal extent of the Columbia River Basalt Group

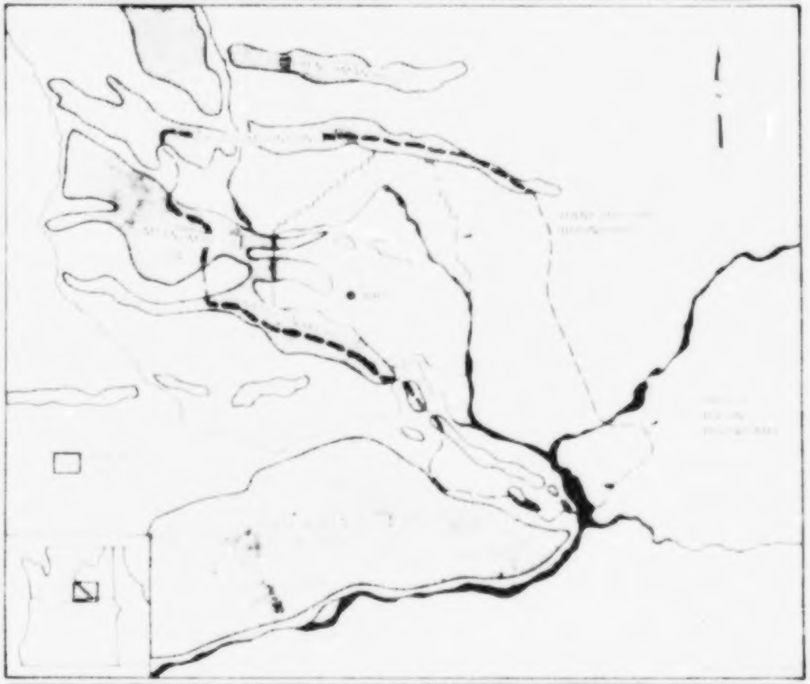


Figure 2 Location of the Pasco Basin, Hanford Site and RRL.

Technical Criteria

Technical criteria with respect to ground-water monitoring strategy development include factors related to site conditions, facility design, and source characteristics. Data related to these factors are generally essential in determining and evaluating logical alternatives for the ground-water monitoring system design. In other instances, trade-offs must be made between the amount of time and effort used in collecting design-basis data and the level of risk and uncertainty acceptable.

Site conditions include [illegible] characteristics of the [illegible] encountered [illegible] the geologic and hydrologic environments. Site conditions [illegible] characterized [illegible] all but [illegible] detail tends [illegible] increasing radius [illegible] these [illegible] are [illegible] That [illegible] field [illegible]

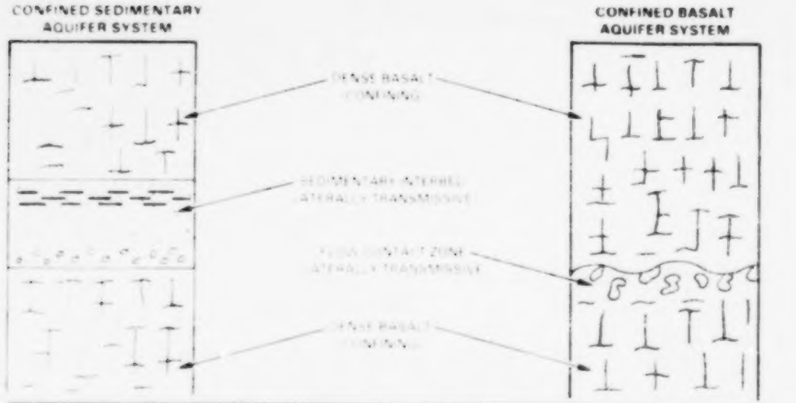

CONFINED SEDIMENTARY AQUIFER SYSTEM
CONFINED BASALT AQUIFER SYSTEM
DENSE BASALT CONFINING
SEDIMENTARY INTERBED LATERALLY TRANSMISSIVE
FLOW CONTACT ZONE LATERALLY TRANSMISSIVE
DENSE BASALT CONFINING

Figure 3 Schematic examples of confined aquifer systems encountered within the Columbia River Basalt

rock which would be directly affected by repository construction and operation, whereas far field refers to the zone beyond the near field. In order to design a ground-water monitoring system it is essential to have a conceptual understanding of both near field and far field hydrogeologic environments. Of principal interest are those factors which act to create flow potential (for example, hydraulic head distributions), direct or bound existing or probable flow paths (for example, geologic situations, basalt flow geometry, stratigraphy), or affect contaminant transport (for example, geochemical environment or flow path). Briefly, the conceptual hydrogeologic system of the Columbia River basalt within the Pasco Basin may be described as occupying a thick, multilayered flood basalt sequence overlain by glaciofluviatile and other sediments (Myers, Price et al. 1979 and Gephart et al. 1979). The sedimentary overburden materials support unconfined and semiconfined ground-water flow systems, whereas lateral flow of confined ground water occurs along certain basalt flow contact zones ("interflows") and within portions of interbedded sedimentary horizons. Thick, dense basalt flow interiors and fine sediments within the sedimentary interbeds comprise confining horizons between zones of lateral ground-water flow (Figure 3). Structures, shear zones and lateral facies can have an apparent, but undetermined, influence on anisotropy. Based on current conceptual understanding of the site hydrogeology, the following considerations have been deemed salient to ground-water monitoring strategy development:

- Multiple ground-water flow systems in a subparallel configuration.
- Anisotropy and lateral heterogeneity.
- Areal variances, possibly due to structural influence.
- Discrete hydrochemical properties associated with certain ground-water flow systems.

These flow system characteristics can have a direct influence on the number of zones to be monitored, the three-dimensional configuration of monitoring sites, and the selection of parameters to be monitored, and must, therefore, be accounted for in the design of the ground-water monitoring strategy.

Similarly, facility design considerations are essential. The current concept of the repository centers around a series of tunnels and adits with an areal extent within the subsurface of, perhaps, a few square kilometers. The underground workings would have a thickness of 12.5 m (~40 ft) and would be situated within the dense interior of a basalt flow having a total thickness of 60 to 75 m (200 to 250 ft). The facility would be situated at a depth of about 1,100 m (3,700 ft) below the land surface. Access would be achieved via several vertical shafts having diameters up to 6.7 m (22 ft). On completion of facility operation, requirements call for complete backfilling of the underground workings and sealing and backfilling of shafts. Key items associated with the repository configuration which may affect ground-water monitoring strategy development include:

- Potential for either point or areal source contamination.
- Significant depth from the surface.
- Vertical access to the repository and principal ground-water flow paths.
- Presence of backfill materials and possible comparative use of these materials as a tracer source.
- Extremely low permeability of the host rock.

Finally, consideration must be given to source (or waste package) characteristics. High-level radioactive waste (HLW) generally differs significantly in character from other hazardous wastes. For example, HLW is capable of generating significant quantities of heat on disposal and has determinable rates of decay to isotopically stable forms (assuming waste composition is known). Typically, HLW consists of a variety of radioactive isotopes each having characteristic radioactive decay, chemical, and sorptive properties. Current NRC policy provides that the engineered HLW package must contain all radionuclides for the first 1,000 years following facility decommissioning. This concept alone poses a significant criterion with respect to ground-water monitoring. The proposed NRC rules also specify requirements for waste package design verification and implied monitoring. The BWIP ground-water monitoring strategy, however, involves only the monitoring of the ground-water system and not engineered structures. A repository monitoring strategy is being developed independently. Thus, a further consideration is the coordination of all the various project monitoring activities into a single strategy. It should also be noted that, in contrast to surface impoundments and near-surface facilities, the path of interest with respect to contaminant excursion from the facility is generally from great depths toward the surface, and, in contrast with deep well injection, the possible source of contaminant release may be areally extensive. For further details regarding waste and engineered barrier concepts, the reader is referred to Smith et al. (1980).

Practical Considerations

Beyond the regulatory and technical criteria noted above, there exist various practical considerations which are also important to the development of a ground-water monitoring strategy. These considerations include:

- Project phasing and scheduling.
- Drilling and borehole constraints.
- Applicability of existing data and ongoing data acquisition activities.
- Technological limitations.
- Determination of acceptable uncertainty levels.

Project phasing and scheduling implies a critical demand on ground-water monitoring strategy development. This is particularly evident in considering the time variant nature of certain hydrologic parameters (e.g., hydraulic potential distributions). An example of a recent iteration of BWIP phasing and scheduling is given in Figure 4. Accounting for the various ground-water monitoring requirements of the proposed NRC rules, the ground-water monitoring strategy must include component systems which provide for the purposes identified from a review of NRC's proposed rules (10 CFR 60).

If baseline monitoring requires sufficient sampling to establish seasonal and long term trends, well in advance of significant site disturbances which could destroy or significantly change the baseline, an appropriate lead time must clearly be scheduled in order to acquire the baseline data. It is thus appropriate to establish a critical path as part of the ground-water monitoring strategy and to integrate this planning into the program schedule. A further consideration involves the hydraulically "tight" character of the repository host rock. These conditions may even preclude the use of conventional monitoring methods to validate predictive models. It may, therefore, be preferable to rely on simulation experiments, such as tracers, to achieve the required level of validation.

If ground-water monitoring is to be achieved by means of boreholes, it is obvious that drilling considerations will be important. Typical considerations include:

- Can drilling activities for ground-water monitoring sites be coordinated to meet other data gathering needs (e.g., better structural or stratigraphic definition, hydrologic testing, etc.)?
- Can existing boreholes be rehabilitated for monitoring purposes?
- At what point would the number of boreholes jeopardize the system performance?
- How can drilling costs be best optimized?

Geologic repositories are probably unique among waste disposal facilities in that the level of geotechnical characterization is quite intense prior to site selection. These characterization activities continue throughout a good portion of the project's operational life. As a result, several boreholes may be present within a given radius of the site at the outset of the monitoring effort. The distribution of various bore

Figure 4 Basalt Repository Phase Schedule

Figure 5 Location of basalt test wells and current hydrologic monitoring sites

holes which have been used by the BWIP for area-wide geologic and hydrologic characterization of the basalts at Hanford is shown in Figure 5. Despite the distribution of possible monitoring sites, the potential for use of these boreholes remains limited because of various design constraints. Most test holes are "NX-type" (7.7-cm/3.032-in. diameter) open core holes which are cased through the basalt overburden. This relatively small-diameter design limits options for converting these existing test holes into multiple-zone monitoring wells. The hydrostratigraphic distribution of these boreholes may, likewise, pose a constraint. Certain boreholes, despite an unfavorably small diameter, might be designed and reconstructed to provide data on individual zones. An efficiently conceived drilling program must account for needs implied by the monitoring strategy. An ideal drilling plan for a multifaceted project, such as the BWIP, should account for borehole needs within a comprehensive plan. The drilling plan should allow for the gathering of time-invariant data (e.g., transmissivity, storativity, stratigraphy, etc.) and the performance of various experiments, while minimizing the number of drilling sites. As appropriate, the boreholes drilled for these purposes could be converted to monitoring wells on completion of testing.

In some instances it may be possible to use available data from historical records to satisfy certain parameter baseline descriptions. Similarly, certain data may be useful in providing projections based on trends. Of particular interest would be data which can correlate statistically with regard to time-dependent parameters. For example, in establishing long-term trends for potentiometric levels in deep ground-water zones, it may be sufficient to monitor at the site only for a time sufficient to show that there is a hydraulic relationship between the data gathered at the site and the historical record for a well situated at some greater radius. Historical records for the unconfined and uppermost confined ground-water systems at the Hanford Site are probably sufficiently complete to represent an excellent baseline for the RRL.

The ground-water monitoring strategy may also be constrained by various technological considerations related to design and instrument capabilities. Perhaps the most important factor, borehole well diameter, has already been eluded to. This is particularly a concern in isolating multiple zones in deep boreholes. Multiple piezometer suites can be constructed, but the borehole diameter increases with each additional piezometer required. Further, increasing well diameter tends to increase the radius of disturbance. Considering that the purpose of a monitoring well is generally to observe the effects of a system modification or disturbance, the well itself should, therefore, minimize disruption of the system to the extent possible. Well diameter also may constrain the ability to collect samples from the zones being monitored. Although it may be possible to collect samples from small-diameter (3.2-cm/1.25-in. i.d.) tubes via gas-lifting or surface-manipulated, piston-type pumps, the ability to properly develop sampling zones is severely limited. An alternative consideration might be multiple-port installation. Other examples of technological constraints include the logistics of measuring parameters in very deep wells and availability of data on instrument longevity.

Finally, with regard to acceptable levels of uncertainty, a variety of trade-offs might be considered. One foremost trade-off is that of the number of monitoring wells required to obtain an acceptable level of risk with respect to contaminant detection. For a given configuration of monitoring wells, the greater the number of wells, the lower the risk of nondetection. At some point, however, risk mitigation due to each additional well would become insignificant and other risks, such as increasing the number of vertical pathways from the repository to the surface, begin to increase. Of course, the ultimate test to what constitutes an acceptable level of uncertainty and risk in this regard is that of public opinion. The role of scientists and engineers associated with a project such as the BWIP can only be to characterize and quantify the risks and to provide appropriate recommendations and/or designs when a decision is made.

Conclusions

In recent years, HLW disposal technology has helped to pioneer applied research in various aspects of hydrogeology (including low-permeability testing, adaptation of oil-field techniques to ground-water studies, ground-water tracers, etc.) The need for this research stemmed largely from the previous lack of experience and interest in the type of hydrologic environment favorable to the geologic repository concept. In other words, hydrologists have been concerned primarily with zones of high ground-water quality and productivity, whereas the objective of subsurface waste isolation is to avoid such zones. Even underground waste injection requires that the target hydrogeologic unit have threshold hydraulic conductivity. The Resource Conservation and Recovery Act (RCRA) has spurred a renewed interest in monitoring ground-water systems and strategies. However, RCRA-type facilities tend to be sited within near-surface environments, and the costs and technology for conducting ground-water monitoring for such facilities are generally less extensive. It is, nevertheless, reasonable to expect that an exchange of information regarding these two technologies might benefit those involved in the system and strategy design of each.

For HLW repositories, ground-water monitoring is an explicit regulatory requirement. It is clear that ground-water monitoring is intended to fulfill several purposes critical to various project phases. This implies a strong need for those involved with project planning to devise a comprehensive strategy for ground-water monitoring. This strategy should encompass all considerations focal to ground-water monitoring as they correspond to respective project phasing.

Despite the obvious differences between RCRA-type and HLW repository-type ground-water monitoring, a continuing information exchange between technicians should be mutually beneficial.

Biographical Sketches

Leo S. Leonhart graduated from Youngstown State University in 1970 with a B.S. in geology. He earned an M.S. in natural resources from Ohio State University in 1971 and a Ph.D. in watershed hydrology from the University of Arizona in 1978. He was employed by the U.S. Environmental Protection Agency from 1974 through 1978, where he investigated the environmental effects of various energy, resource-extraction, and water-resource proposals. He joined Rockwell Hanford Operations in 1978 as a senior hydrologist and is currently responsible for the Basalt Waste Isolation Project's regional hydrology and ground-water monitoring activities. In addition to membership in

NWWA, he belongs to AGU, AIME and ASTM and is a certified professional geologist in the state of Indiana and by AIPG.

Frank A. DeLuca is presently the manager of the Basalt Waste Isolation Project's Hydrology Group, where he has control over the project drilling, field testing and hydrologic modeling effort. Prior to joining Rockwell, he was a senior scientist with Geraghty and Miller Inc. (1966-1979) and was with the Federal Water Pollution Administration and the USGS, Water Resource Division (1959-1966). DeLuca is a certified professional geologist by AIPG and is a registered professional geologist in the states of Delaware, Georgia and Oregon. He received his B.A. in geology from Hofstra University in 1959.

N. Thomas Sheahan is chief geologist and senior project manager for Brown and Caldwell Consulting Engineers, Pasadena, California. Prior to joining Brown and Caldwell, he was a research engineer and hydrogeologist for Layne and Bowler Inc., Memphis, Tennessee (1965-1972) and an engineering geologist for the U.S. Army Corps of Engineers and Woodward-Clyde Consultants (1960-1965). Sheahan received his B.S. in geology from the University of Missouri in 1960 and completed graduate studies in advanced ground-water hydrology at M.I.T. in 1967. He also earned the degree of Juris Doctor in 1978 from the University of LaVerne, College of Law. Sheahan is a certified professional geologist by AIPG, a registered professional geologist in the states of California and Oregon, and a licensed attorney at law in California.

Lawrence M. West has been a project geologist for Brown and Caldwell Consulting Engineers since 1978. Prior to this, he was a geohydrologist with the ground water division, Oregon Department of Water Resources, a geologist with Rittenhouse-Zeman and Associates, a ground-water geologist with Stang Hydronics Inc. and a geologist with Woodward-Clyde Consultants (1972-1978). West received his B.S. in geology from California State University at Northridge in 1972 and has done graduate work at California State University at Los Angeles and at Fullerton.

References

Myers, C.W., S.M. Price, J.A. Caggiano, M.P. Cochran, W.J. Czimer, N.J. Davidson, R.C. Edwards, K.R. Fecht, G.E. Holmes, M.G. Jones, J.R. Kunk, R.D. Landon, R.K. Ledgerwood, J.T. Lillie, P.E. Long, T.H. Mitchell, E.H. Price, S.P. Reidel and A.M. Tallman. 1979. Geologic Studies of the Columbia Plateau, A Status Report. RHO-BWI-ST-4. Rockwell Hanford Operations, Richland, Washington.

Gephart, R.E., R.C. Arnett, R.G. Baca, L.S. Leonhart and F.A. Spane Jr. 1979. Hydrologic Studies Within the Columbia Plateau, Washington: An Integration of Current Knowledge. RHO-BWI-ST-5. Rockwell Hanford Operations, Richland, Washington.

Smith, M.J., G.J. Anttonen, G.S. Barney, W.E. Coons, F.N. Hodges, R.G. Johnston, J.D. Kaser, R.M. Manabe, S.C. McCarel, E.L. Moore, A.F. Noonan, J.E. O'Rourke, W.W. Schulz, C.L. Taylor, B.J. Wood and M.I. Wood. 1980. Engineered Barrier Development for a Nuclear Waste Repository in Basalt: An Integration of Current Knowledge. RHO-BWI-ST-7. Rockwell Hanford Operations, Richland, Washington.

Dup

THE RADIONUCLIDE GROUND-WATER MONITORING PROGRAM FOR THE SEPARATIONS AREA, HANFORD SITE, WASHINGTON STATE

By Michael J. Graham

Introduction

Overview

The U.S. Department of Energy (DOE) Hanford Site has served as an integrated nuclear facility since 1943. The site occupies 1,476 square kilometers in semi-arid, south-central Washington State. An extensive, unconfined aquifer system is contained within glaciofluvial sands and gravels and silts and gravels of the Ringold Formation. The bottom of the aquifer is the basalt bedrock surface of the Columbia River Basalt Group. The aquifer at the Hanford Site is more than 70 meters thick in some areas and is absent along the flanks of basaltic structures.

The Separations Area is located near the center of the Hanford Site. The area contains uranium fuels processing facilities, plutonium separation facilities and the major radioactive waste storage and disposal facilities. From the plutonium separation processes, low-level radioactive wastes are discharged to the ground by means of surface and subsurface structures.

As part of the surveillance and maintenance program at Hanford, an extensive ground-water monitoring network is maintained. The purposes of this ground-water monitoring program are:

- to assess the performance of disposal facilities;
- to determine the impact of these liquid waste discharges on the quality and flow of ground water;
- to provide data input for the calibration of hydrologic models.

The monitoring network consists of 108 wells. Samples from these wells are collected at frequencies ranging from monthly to semi-annually. Samples are selectively analyzed for radioactive and nonradioactive constituents. Data are interpreted by comparing the analytical results with established histories and concentration guides; contamination plume maps are generated yearly. All aspects of the monitoring program, from design to data interpretation, are founded on an understanding of the ground-water flow system.

Background

Rockwell Hanford Operations (Rockwell) is a prime contractor for waste management to DOE at Hanford. The Separations Area, located near the center of the Hanford Site, contains the major radioactive waste storage and liquid waste disposal facilities (see Figure 1).

As part of the waste management effort, a comprehensive ground-water monitoring network is established for the Separations Area. Under this monitoring program, water samples are collected at various time intervals from over 100 wells located near liquid-waste disposal facilities. These samples are analyzed for radionuclide and chemical contamination.

In addition to the Separations Area ground-water monitoring program, Pacific Northwest Laboratory (PNL) maintains the Hanford ground-water monitoring program. The purpose of the Hanford ground-water monitoring program is to estimate and evaluate the impact of unconfined ground-water contamination from the Hanford Site on the general public. Ground-water sampling and analysis for the two programs are coordinated and data are actively exchanged.

Purpose and Objectives

The purpose of this paper is to present an overview of the hydrogeology and the radionuclide ground-water monitoring program of the Separations Area at Hanford. The objectives of the monitoring program are to:

- assess the performance of the low-level liquid-waste disposal facilities;
- determine the impact of liquid-waste discharge on the quality and flow of ground water;
- provide data for input to ground-water flow and transport predictive models.

Waste Disposal and Storage Facilities

High-level Waste Storage

High-level radioactive waste produced in the chemical processing of irradiated uranium fuels is stored in large underground tanks. High-level radioactive wastes are defined as those wastes produced from first-cycle chemical processes designed to separate the highly active fission products from the residual useful fuels (Macbeth, et al., 1978). The acidic wastes are neutralized for storage, resulting in salt, sludge and a supernatant liquid. Between 1952 and 1958, depleted uranium was recovered from these tanks. Since 1952, waste volume reduction by evaporation has been the primary objective of high-level waste management at Hanford.

Initially, 64 single-shell tanks were constructed for high-level waste storage. An additional 85 single-shell tanks were constructed between 1946 and 1964. The single-shell tanks have capacities of 200,000 or 3,800,000 liters (55,000 or 1,000,000 gallons). Starting in the late sixties, storage tanks for high-level waste were built with two carbon steel liners (double-shell tanks). All single-shell tanks have been stabilized by removing the supernatant liquid. These high-level liquid wastes are now stored in double-shell tanks.

Low-level Liquid Waste Disposal

Beginning in the mid-forties, small volumes of selected low-level radioactive wastes were discharged to the ground by means of subsurface structures. The rate of low-level liquid waste discharged to the ground increased steadily and reached a peak in 1955. Thereafter, the quantity of low-level wastes discharged has decreased as ground disposal of radionuclides was minimized through process improvements and shutdown of plants in response to cutbacks in defense needs.

There have been 195 subsurface disposal facilities constructed in the Separations Area. The following types of facilities have been utilized for liquid-waste disposal at Hanford:

- **Cribs**—subsurface liquid dispersal systems, used for low-level disposal process, condensate and lab wastes
- **Trenches**—unlined excavations, generally used for short periods on a specific retention basis for the disposal of high-salt waste or waste containing complexed radionuclides
- **French drains**—covered or buried gravel-filled encasements with open bottoms, used for the disposal of small volumes of low-level waste
- **Reverse wells**—buried or covered, encased, drilled holes with the lower end perforated or open, used for the disposal of process waste. Reverse wells were removed from service because they plugged easily and introduced waste at or near the ground-water surface.

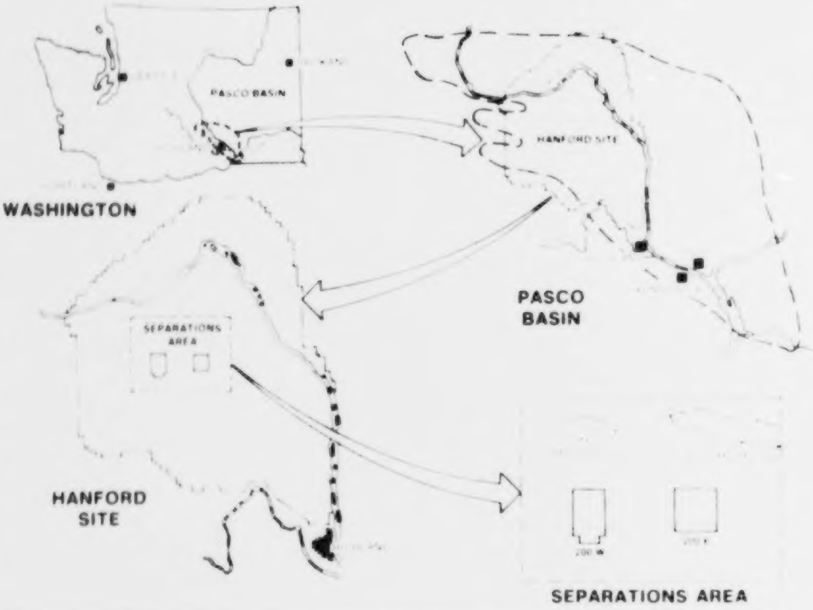


Figure 1 Separations Area Location Map

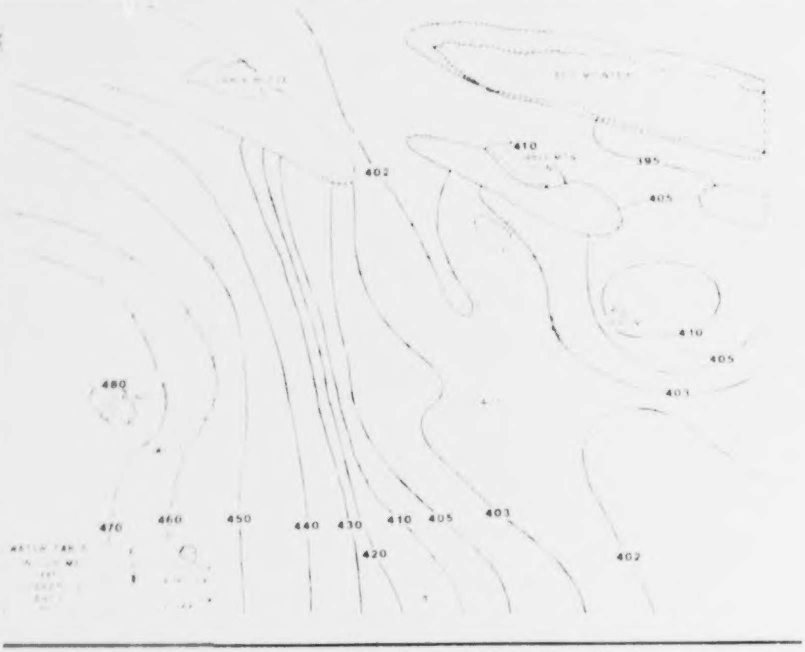


Figure 2 Water-table map of the Separations Area

The large volumes of cooling water and steam condensate from chemical processing facilities and evaporator-crystallizers are discharged to surface ponds and ditches. Normally, the radionuclide concentrations are below DOE maximum permissible concentration guides for an uncontrolled area, but occasional nonroutine releases of higher-level wastes have occurred. Ponds are natural or diked surface depressions which allow the liquid effluent to percolate into the underlying sediment. The major disposal ponds are U-Pond, B-Pond and Gable Mountain Pond (Figure 2). Ditches are unlined excavations used for conveying the low-level liquid waste to the ponds. Through 1980, more than 1 x 10^{10} liters (2 x 10^{9} gallons) of liquid waste had been discharged to the ground since start up.

Solid Waste Burial

About 1.9 x 10^{5} cubic meters (6.7 x 10^{5} cubic feet) of contaminated solids have been buried in the Separations Area since the start of chemical processing operations (Isaacson and Brown, 1978). These wastes consist of dry waste (composed of soiled clothing, laboratory supplies, tools, etc.) packaged in cardboard, wood or metal containers; and industrial waste (primarily items of failed process equipment) packaged in plastic shrouds, wood, metal or concrete boxes.

Geology

The Hanford Site is located within the Pasco Basin, a structural and topographic basin in south-central Washington State. The boundaries of the Pasco Basin are defined by anticlinal structures of basalt. The stratigraphy beneath the Separations Area consists of the Columbia River basalt group, composed of the Grande Ronde, Wanapum and Saddle Mountains Formations (Ledgerwood, et al., 1978). The Saddle Mountains basalt within the area consists of four basalt flows separated by interbedded sediments. The individual basalt members are, in ascending order, the Umatilla, Esquatzel, Pomona and Elephant Mountain. The major interbedded sediments are the Mabton, Cold Creek, Selah and Rattlesnake Ridge members which are part of the Ellensburg Formation. These interbeds and basalt interflow zones form an extensive confined aquifer system.

The Ringold Formation overlies the Elephant Mountain member, except in the northeast corner of the Separations Area where these sediments have been removed by erosion. The Ringold Formation can generally be divided into four informal units on the basis of texture: the sand and gravel unit of the basal Ringold; the clay, silt and fine sand with lenses of gravel of the lower Ringold; the occasionally cemented sand and gravel unit of the middle Ringold and the silt and fine sand unit of the upper Ringold (Tallman, et al., 1979).

The glaciofluvial sediments, informally named the Hanford Formation, are deposited on the Columbia River basalt group and Ringold Formation. These sediments can be divided into the coarser sand and gravel units, which are referred to as the Pasco gravels, and the finer sand and silt units, called the Touchet beds. Loess deposits veneer most of the Separations Area; sand dunes are present in the southern portion of the area.

Table 1
Ranges of Hydraulic Properties of the Unconfined Aquifer in the Separations Area

Interval Tested	Hydraulic Conductivity	
	m/day	ft/day
Hanford Formation	600-3000	2,000-10,000
Middle Ringold Unit	3-70	9-230
Lower Ringold Unit	0.1-3.6	1-12
Region	**Transmissivity**	
	m^2/day	ft^2/day
200 West	28-500	300-5,400
200 East	.5-12,500	5-135,000

Climate

The climate at Hanford is dominated by movement of storm fronts from the Pacific Ocean eastward over the Cascade Mountains. Summers are sunny, warm and dry; winters are relatively mild.

At the Hanford Meteorology Station, located in the Separations Area, average annual precipitation is 16 centimeters (6.25 inches). Over 30 percent of this precipitation occurs during November, December and January; roughly 10 percent occurs in July, August and September. Almost 50 percent of the moisture falling during the months of December through February is in the form of snow.

Hydrology of the Separations Area

Occurrence of Ground Water

The aquifer impacted by liquid waste disposal at Hanford is the unconfined aquifer. The unconfined aquifer is contained within the Ringold and Hanford Formations. The bottom of the aquifer is the basalt surface or, in some areas, the clay zones of the lower member of the Ringold Formation. Basalt subcrops and outcrops extend above the water table in the north and northeast portions of the area and act as barriers to lateral ground-water movement (Figure 2). Depth to ground water ranges from 55 to 100 meters (180 to 333 feet); the aquifer reaches a maximum thickness of 70 meters (230 feet).

Aquifer Properties

Large differences in hydraulic conductivity are evident between the Hanford Formation and the Ringold Formation. The Hanford Formation consists of loose, permeable sands and gravels, whereas the middle Ringold contains more fine grained material, is more compact and is semi-consolidated in places. The Hanford Formation normally exhibits a hydraulic conductivity of between 600 and 3,000 meters/day (2,000 to 10,000 feet/day); the middle Ringold averages 30 meters/day (100 feet/day) (Graham et al. 1981).

Aquifer properties can also be delineated based on location (Table 1). In general, the 200 East Area unconfined system displays a higher transmissivity than the 200 West Area. The higher values are attributed to a thickening of the aquifer (due to a drop in the basalt surface) east of 200 West Area and the presence of the aquifer in the more conductive Hanford Formation in 200 East Area.

Within the unconfined aquifer, storativity ranges from 0.002 to 0.07, the lower value associated with the lower Ringold Formation and the higher value associated with the Hanford Formation. Specific yield has been estimated at two locations, where values of 0.15 and 0.18 were calculated.

The unconfined aquifer is highly anisotropic, with horizontal to vertical hydraulic conductivity ratios of 13 to 16. This high value gives an indication of preferred horizontal flow through the unconsolidated sediments rather than vertical flow through the aquifer. Vertical flow occurs in the unconfined aquifer but is inhibited by clay, silt, fine sand and cemented horizons.

Flow Dynamics

The total volume of water added to the aquifer per year by liquid waste disposal to the ground is approximately 10 times the natural inflow to the area from precipitation and irrigation waters infiltrating in Cold Creek Valley to the west (Graham, et al., 1981). Percolating water from U-Pond and B-Pond (Figure 2) has resulted in the formation of large mounds under these sites. The water table has risen 26 meters (85 feet) under U-Pond and nine meters (30 feet) under B-Pond compared to pre-Hanford conditions (Newcomb, et al., 1972).

Ground-water flow patterns are dominated by these ground-water mounds. The movement of ground water from west of U-Pond is redirected around the mound. Flow from U-Pond is primarily toward the east; gradients are extremely steep, greater than 10 meters/kilometer (50 feet/mile). Between the 200 East and 200 West Areas the gradient abruptly flattens out. This is attributed to an increase in the hydraulic conductivity of the sediments and a drop in the basalt surface which results in an increase of nearly 40 percent in the thickness of the saturated zone. This essentially causes a "pooling" of the ground water in this area. The flow system in East Area is complex due to the change in aquifer thickness and hydraulic properties, the influence of B-Pond and the complicated structures of the Gable Mountain-Gable Butte area. Flow out of the 200 East Area is to the north, through Gable Gap, then south along a ground-water trench defined by the 402 foot contour. Flow from B-Pond is radially outward to the east and combines with the flow paths out of 200 East Area. Discharge from the unconfined aquifer is principally to the Columbia River, with minor discharge to the Yakima River.

Separations Area Ground-Water Monitoring Program

The monitoring program for assessing radionuclide ground-water contamination under low-level radioactive disposal facilities is based upon the understanding of the hydrology of this area. Knowledge of the ground-water flow system predicates well location, well completion and sampling frequency.

Well Network

There are 108 water-quality monitoring wells in the Separations Area ground-water monitoring network. The wells are either 15 cm (6 in.) or 20 cm (8 in.) in diameter; drilling smaller diameter holes has been largely unsuccessful due to the depths to ground water (up to 100 meters [330 feet]) and drilling conditions. The larger casings also allow access with hydrologic testing equipment. Each well is fitted with a one-half horsepower submersible pump. Piezometers are also used to monitor head values and water quality with depth. Most wells are completed in the upper portion of the aquifer.

The wells are on a five-year maintenance program. Under this maintenance program the wells are inspected with a down-hole television camera. Any structural deficiencies in the wells are repaired. The wells are then cleaned and developed.

Sampling

The following criteria are established for sampling frequencies:

- wells monitoring active liquid waste disposal sites are sampled monthly;
- wells monitoring inactive liquid-waste disposal sites which contain contaminants that have a high potential of being remobilized are sampled monthly;
- wells (other than specific sites mentioned above) indicating contamination above background are sampled monthly or quarterly depending upon the level of contamination and the stability of the concentration trends;
- wells yielding samples indicating background contamination are sampled semiannually.

All wells included in the monitoring network are sampled over a sufficient length of time to provide a sound data base for statistical analysis. Standard field procedures are used to collect the ground-water samples. The samples are field prepared for subsequent analysis.

Analysis

The analytical schedules are based upon the inventory of the sites being monitored and the concentration histories of the wells. An inventory of input and decayed values of radioactive liquid wastes discharged to the ground is maintained for each disposal site. Concentration histories starting in the 1950s exist for many of the wells. The samples are selectively analyzed for the following radionuclides: total alpha, total beta, cobalt-60, ruthenium-106, cesium-137, strontium-90, tritium and uranium. Non-routine analyses include technetium-99, neptunium-237 and plutonium-239.

Total alpha and total beta are used as general indicators of radionuclide contamination. When total alpha or total beta are detected in significant concentrations, the individual alpha and beta emitters are identified. Nitrate is also present in the process waste streams and has been used routinely at Hanford to define contamination migration in the ground water.

Data Interpretation, Reporting and Storage

Data are interpreted in a timely manner, usually the same day the results are obtained from the laboratory. The data are interpreted to assess the performance of the active liquid-waste disposal facilities and to determine general contamination patterns. In assessing the performance of the disposal sites, the data are compared with the DOE concentration guides for water in an uncontrolled area. If the ground-water contamination from an active site exceeds 10 percent of the concentration guides for the long lived radionuclides, the disposal facility is removed from service. This standard of 10 percent is adopted to assure that the concentration guides are not exceeded by subsequent drainage after the facility is removed from service.

Tritium contamination plumes (Figure 3) show general patterns of ground-water contamination, which is primarily the result of past disposal practices. The tritium contamination has migrated away from the various liquid-waste disposal facilities in a general west to east direction along major flow lines. The tritium contamination from the 200 West Area has not migrated as far and appears to have dispersed less than contamination originating from the 200 East Area. In the 200 East Area, the aquifer is present in the Hanford Formation sediments, which display higher hydraulic conductivities and lower sorption coefficients than those of the Ringold Formation, which dominates the aquifer in the 200 West Area. Analysis of data from the ground-water monitoring network indicates that contamination levels in ground water migrating out of the Separations Area are below DOE concentration guides for water in an uncontrolled area.

The concentration histories for tritium and total beta in well 299-W19-5 and monitoring crib 216-S-23 are shown in Figure 4. Discharge to the crib, which was placed in use in January 1969, reached a high in 1970 and dropped off sharply in 1971 until discharge was terminated and the crib was removed from service in July 1972. Total beta, which is primarily ruthenium-106 (half-life 367 days), rose sharply to a

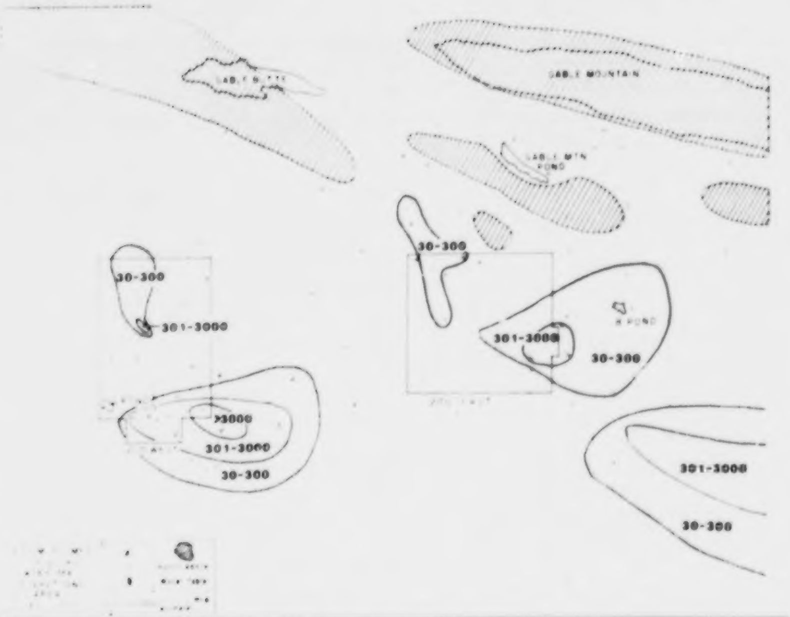


Figure 3. Tritium plumes within the Separations Area

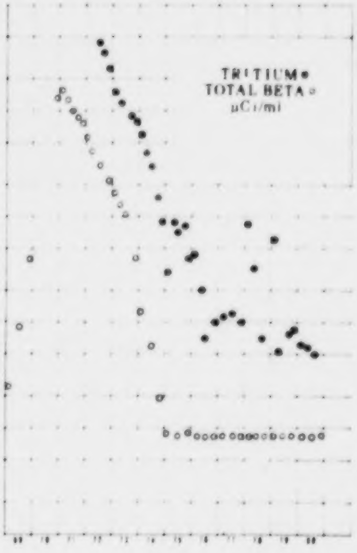


Figure 4. Concentration histories of total beta and tritium in well 299-W19-5

peak of 1.3×10^{4} µCi/ml in early 1971. Total beta and tritium concentrations dropped sharply due to radioactive decay and dispersion as discharge to the crib was decreased and eventually terminated.

The data from the Separations Area ground-water monitoring network are stored on the CIRMIS (Comprehensive Information Retrieval and Model Input Sequence) computer data-base system. Reports are issued quarterly. An annual report is also written and included in the annual environmental surveillance report for the Separations Area.

Quality Assurance

Quality assurance is an integral part of the Separations Area ground-water monitoring program. The quality assurance program consists of internal and external audits. For each aspect of the routine program and every party involved in the program, internal audits are performed annually, as a series of independent audits or a single comprehensive audit. These audits address the following:

- documentation of well condition and maintenance;
- compliance with sampling and analytical procedures;
- precision and accuracy of analytical results;
- evidence of data review, interpretation and response, and
- evidence of internal audit reports.

The external audits, which are performed by Rockwell Quality Assurance Department, address compliance with internal quality assurance programs.

References

Graham, M.J., Hall, M.D., Strait, S.R. and Brown, W.R., 1981, "Hydrology of the Separations Area," RHO-ST-42, Rockwell Hanford Operations, Richland, Washington.

Isaacson, R.E. and Brown, D.J., 1976, "Environmental Assessment Related to Hanford Radioactive Waste Burial," RHO-SA-36, Rockwell Hanford Operations, Richland, Washington.

Ledgerwood, R.K., Myers, C.W. and Cross, R.W., 1978, "Pasco Basin Stratigraphic Nomenclature," RHO-BWI-LD-1, Rockwell Hanford Operations, Richland, Washington.

Macbeth, P.J., Christensen, D.E., Thamer, B.J. and Wehmann, G., 1978, "Screening of Alternative Methods for the Disposal of Low-level Radioactive Wastes," Ford, Bacon and Davis, Utah Inc. NTIS, NUREG/CF-0308.

Newcomb, R.C., Strand, J.R. and Frank, F.J., 1972, "Geology and Ground-Water Characteristics of the Hanford Reservation of the U.S. Atomic Energy Commission, Washington," U.S. Geological Survey Professional Paper 717.

Tallman, A.M., Fecht, K.R., Marratt, M.C. and Last, G.V., 1979, "Geology of the Separations Area, Hanford Site, South-Central Washington," RHO-ST-23, Rockwell Hanford Operations, Richland, Washington.

Biographical Sketch

***Michael J. Graham** is a senior hydrologist for Rockwell Hanford Operations, in charge of Rockwell's ground-water monitoring network. He holds an M.S. in geology (specializing in hydrology) from Indiana University and has completed course work toward his Ph.D. in geology, also at Indiana University.*

GROUND-WATER MONITORING TECHNIQUES FOR ARID ZONE HAZARDOUS WASTE DISPOSAL SITES

by Robert F. Kaufmann, T. Alan Gleason, R. Brian Ellwood and Gerald P. Lindsey

Recent field studies conducted at an existing solid and liquid hazardous waste disposal site in California and at a wastewater treatment facility under construction in Arizona necessitated use of a variety of monitoring techniques due to contrasts in hydrogeologic conditions, waste sources and pollutant transport pathways. Also relevant is the fact that the California facility was sited and has been operating under less stringent rules than would now prevail, whereas the facility in Arizona reflects state-of-the-art design and construction. Discussion of site-specific conditions and findings is deliberately avoided to ensure confidentiality and to focus attention on the monitoring techniques. The discussion which follows highlights both water quality and ground-water flow system monitoring techniques.

Although not directly pertinent to the subject of this paper, the objectives common to both investigations are provided to assist in orienting the reader:

- Define ground-water quality in the aquifers beneath and adjacent to the wastes.
- Delineate the present and/or projected extent of subsurface contamination.
- Define the hydraulic conductivity of the materials beneath disposal areas.
- Define the elements of ground-water monitoring and protection plans responsive to RCRA and state requirements.

Monitoring Rationale

Outlined in Table 1 are the major elements of the ground-water monitoring investigations undertaken at the California and Arizona sites. Those items with an asterisk are discussed in the sections to follow.

At both locations, the preliminary phases of investigation included a review of pertinent literature, a field reconnaissance study and an inventory of nearby active, inactive or abandoned water wells. This was followed by the application of numerous drilling, sampling and analytical techniques which are the focus of this paper.

At the Arizona facility, a wastewater pond complex is present which consists of contiguous wastewater ponds equipped with double liners and a leachate collection system. The basic objective of the investigation for this facility was to design, install and test a ground-water protection system which was both technically and economically sound. Maximum use is made of an existing production well to monitor and intercept seepage from the wastewater ponds, should leakage ever advance farther than the double liners, the leachate collection system and over 300 feet of unsaturated fine-grained desert soils. Two monitoring wells installed downgradient of the ponds complete the well network. Soil resistivity data from depth profiling outside the facility perimeter and from buried electrodes beneath two of the four ponds establish baseline conditions against which any future changes can be compared. Periodic sampling provides additional protection against undetected long-term fluid release. The operational monitoring plan is integrated with normal ground-water pumping cycles to intercept contaminants which might occur at onsite locations.

Monitoring Techniques

Air and Hydraulic Rotary Drilling

Subsurface investigation at the California site involved both air and hydraulic rotary drilling to define stratigraphic variations with depth and water table position. Air rotary drilling proved to be very fast in the poorly cemented sandstone. Bed changes based largely on color variations between the sandstone and claystone units were readily detected. However, the boreholes below the water table were difficult to hold open for proper completion with screened or slotted casing, silica sand backfill, and either cement or bentonite seals installed with a tremie line. Borings made with air rotary equipment were most useful in the initial drilling program in obtaining water-table position and in emplacing suction lysimeters or percolation test holes.

Deeper borings at the California location penetrated to about 50 to 100 feet below the water table and were most successful when drilled with hydraulic rotary equipment and polymer mud. Although setup time for mud pits, pumps, etc., was longer than for air rotary drilling, borings drilled with mud remained fully open for geophysical logging, casing and completion. In particular, sand-pack materials could be readily installed in the screened/slotted intervals. Well development was by means of air and by bailing.

Rotary drilling techniques and polymer muds were used exclusively to install two 400-foot monitoring wells at the Arizona facility. Aside from the usual grab sampling, a wire-line piston sampler was very successfully used about every 40 feet to obtain in-situ samples to confirm cuttings data, to provide reference points for geophysical log interpretation, and to develop accurate grain-size curves for use in screen and gravel-pack design. In order to sample, the drilling string was removed and the sampler was run in on the sand line. Most of the time spent in sampling was in tripping out and in with the drill string. Sample recovery was excellent.

Polymer muds are biodegradable through bacterial action and have variable but limited life, depending on temperature, presence of bacteria in the drilling water, and other factors such as pH. Residual chlorine levels must be maintained in the drilling fluid in accordance with manufacturers specifications. Useful life can be extended by the addition of formalin to retard bacterial action, and by temperature control through shading of the mud pit. Despite the extra degree of care and skill needed on the part of the contractor, use of such muds is warranted because of minimal formation damage. These muds are excellent for geophysical logging and sampling. Special precautions are needed to chlorinate the gravel pack and well water and to overpump the well to eliminate potentially unacceptable levels of coliform bacteria.

Table 1.
Principal Ground-Water Monitoring Elements

Existing Hazardous Waste Landfill

- Definition of Site Geology
  - Literature Review
  - Geologic Mapping
  - Trench Studies to Evaluate Fault Activity
  - Refraction Survey
  - Borehole Geophysical Logging
- Ground-Water Flow System Delineation
  - Piezometers*
  - Sampling Wells*
  - In-situ and Laboratory Permeability Measurements
  - Water Level Measurements
  - Water Well Inventory
- Water Quality
  - Source Term (Waste) Characterization
  - Sampling of Wells, Springs, Seeps*
  - Analysis of Pesticides, Metals, Volatile and Base Neutral Organics
  - Characterization of Natural vs. Introduced Organics in Ground Water*
- Development of Remedial Action Programs
  - Water Saturation Control
  - Digital Pollutant Transport Modeling
  - Long-Term Monitoring Program

New Wastewater Pond Complex

- Baseline Ground Water Conditions
  - Literature Review
  - Water Well Inventory
  - Water Table Configuration
- Ground-Water Monitoring
  - Baseline Resistivity Surveys*
  - Electrode Detection System*
  - Ground-Water Monitoring Wells*
  - Utilization of Existing Pumping Cones
  - Water Sampling

Piezometer and Sampling Well Installation

Piezometers and sampling wells were installed in rotary drilled borings advanced by air and hydraulic rotary rigs, by hollow stem auger, and by driven cylindrical piles. With few exceptions, installation involved a gravel or sand pack and either a bentonite or cement seal. Both pack and seal materials were installed via a tremie line. Pressure grouting was by means of a positive displacement pump. Figures 1 through 3 illustrate the principal construction details of the bored installations. Indication of acceptable sealing of piezometers at the California site is seen in the consistent head measurements decreasing with depth, confirming recharge gradients expected in the highland areas. Recharge is also indicated by development of a water-table mound in the wastes.

Casing materials used included

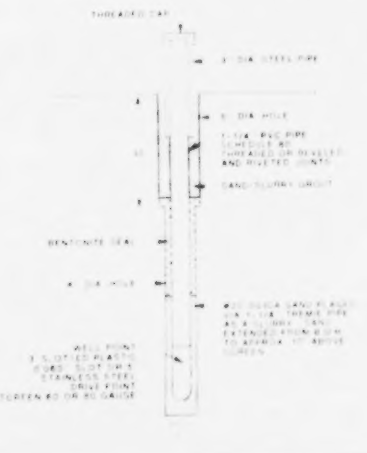


Figure 1. Construction of a typical piezometer.

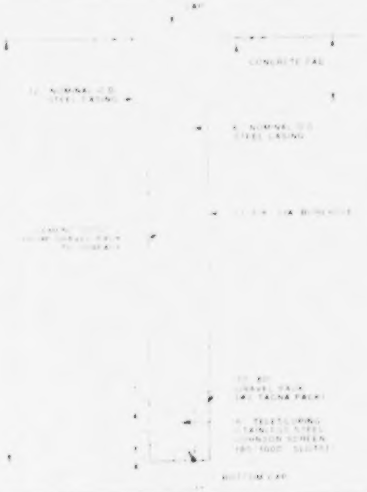


Figure 2. Construction details of a deep sampling well-steel casing.

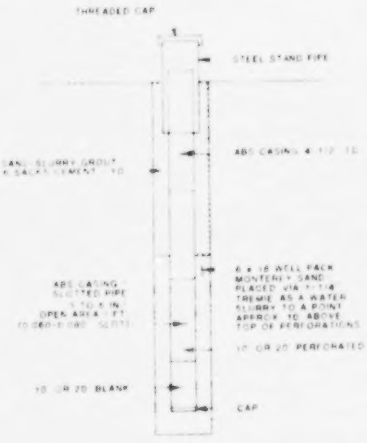


Figure 3. Construction details of a deep sampling well-plastic casing.

1¼-inch steel and PVC pipe, 5-inch ABS pipe with flush screw joints and both 5-inch and 6-inch steel casing with both threaded and welded connections. PVC and ABS pipe was found to be economical considering rapid installation time relative to steel casing. ABS pipe is very durable, but one failure of a slotted section occurred from extreme grout pressure and associated elevated curing temperature. The major disadvantage of ABS and PVC relates to water quality (Scalf, et al., 1981).

Sampling Methods

Soil, soil-moisture, and groundwater samples were collected from borings, suction lysimeters and piezometers or sampling wells, respectively. The discussion which follows focuses on techniques which are not routinely used, as these are considered to be of more interest to the reader.

Soil Sampling

In addition to grab sampling and cuttings, sampling of soil at the California location consisted of coring and Pitcher barrel samplers. A wire-line corer was used to facilitate rapid core recovery, but several problems developed from the mismatch of equipment to subsurface conditions. Coring was done with air for cuttings removal, but condensation around the core barrel and drill stem caused the clay soils to expand, thereby restricting air flow.

Subsequent soil sampling at the California site was with a Pitcher barrel sampler in conjunction with bentonite and polymer drilling fluids to recover consolidated samples immediately above and below the water table. The samples were used to confirm lithologic descriptions available from cuttings and also to perform subsequent analyses for contained organics. The decision to analyze for contaminants was largely based on water-quality data. Use of the Pitcher barrel was limited to relatively shallow rotary wash borings which were advanced with a tri-cone bit and bentonite or polymer drilling fluid.

At the Arizona site a split-spoon sampler in conjunction with a downhole hammer on a wire line was used for soil sampling two monitoring well borings and a pilot hole for a high-capacity production well. The sampler consists of a hammer and drive corer lowered into a boring and operated by alternately raising and lowering the hammer.

Aside from some cross-mixing of cuttings in the immediate bottom of the boring, the resulting samples were fully representative of in-situ conditions. The samples were of considerable value for interpreting electrical and cuttings logs and for sizing of well screens used in one production well and two monitoring wells at the Arizona facility. In addition, the samples could be used for chemical analysis and for developing an improved understanding of the stratigraphic section in the unsaturated zone. In an arid setting, the unsaturated zone is typically rather thick and plays a significant role in the vertical and lateral pattern of contaminant migration. The data obtained are therefore of value in designing ground-water contamination assessment programs as specified in the RCRA regulations.

Well Completion in Hazardous Waste

With regard to the California site, which received both liquid and solid hazardous wastes for over 10 years, initial piezometers and sampling wells were installed in bedrock materials adjacent to the wastes. Projection of static water levels from these wells laterally into the wastes indicated saturation of about a 25-foot thickness of wastes. An additional investigation made to confirm the thickness and saturation of the hazardous waste involved the use of four dual purpose sampling and dewatering wells completed in the wastes or immediately beneath the wastes. The manner in which this was done necessarily had to consider safety in addition to technical and economic practicality. To our knowledge, this was the first drilling **program in California to deliberately penetrate a hazardous waste landfill.**

After considering hollow stem auger, large diameter auger (caisson rig), air and hydraulic rotary, cable tool and driven pile options, the latter was chosen as the best compromise under the given conditions. The method consisted of simultaneously driving two steel cylinder piles, one inside the other, with a 54,000 foot-pound diesel hammer. The piles had inside diameters of 10 inches and 12 inches, wall thicknesses of 0.5 inches, and a combined weight of approximately 177 pounds per lineal foot. The inner pile was approximately six inches shorter than the outer pile, and was fitted with a reinforced flat tip.

After the pile assembly was driven to refusal, the inner pile was withdrawn, allowing completion of 5-inch diameter wells inside the 12-inch pile or, with additional drilling, completion below the piles. Both options were exercised, depending on location and hole conditions. Figure 4 illustrates basic construction features of the pile technique. Drilling below the piles was done at two locations with an air rotary rig, following installation of 8-inch diameter PVC pipe to increase the return of cuttings. Alternately, the boring was advanced with hollow-stem auger (6-inch inside diameter). In the latter instance, the final dual completion featured 1¼-inch steel pipe and screen, sand-packed and sealed in the underlying bedrock and, after withdrawal of the auger, a 5-inch screen and steel casing completion inside the pile.

After installation of the sampling well casing, screen, sand pack and seals, three of the 12-inch diameter

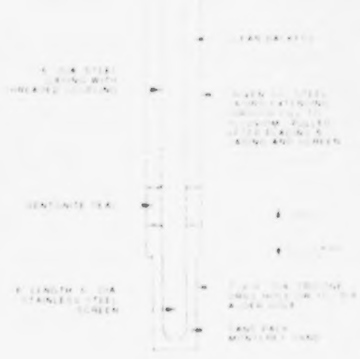


Figure 4. Driven pile sampling and pumping well completion.

Figure 5. Generalized view of submersible pump installation in monitoring well.

cylindrical piles were removed from depths of 34 to 64 feet using a vibratory hammer and truck-mounted crane. At a fourth location, the pile was left in place and the sampling well was completed at the waste-substrata interface which was also the top of the zone of saturation.

Several comments can be made concerning the use of driven piles for the purposes described:

- The method was considered extremely safe insofar as no waste or leachate was removed from the monitoring installation. Routine safety measures for explosive gases were followed when welding or cutting operations were underway.
- Wastes were readily penetrated for the full thickness of 25 to 60 feet at three locations. At the fourth location, metal objects caused pile refusal before the wastes were fully penetrated. Subsequent rotary drilling to penetrate to the base of the wastes was successful but difficult and time consuming.
- Yield of the 5-inch diameter wells, fitted with six feet of stainless well screen was less than 1 gpm in three of the four wells, and was considerably less than expected. Possible causes for this include low yield of the wastes, smearing of the borehole wall as the pile advanced, and/or compaction of the wastes by pile driving or removal.

The pile technique was considered acceptable for the purposes of this study insofar as static water levels in the waste, waste thickness, and leachate quality were confirmed with no adverse safety or environmental impacts.

Water-Quality Monitoring

Ground-Water Sampling Wells

For a variety of reasons, principal ground water sampling wells constructed at both of the sites investigated were a minimum of five inches in diameter. Small-diameter wells were not practical in terms of time needed for well development or sampling and cost effectiveness considering the high cost of labor and chemical analyses. Where possible within budgetary limits, ¼ to ⅓ horsepower pumps were permanently installed, as shown in Figure 5, in six locations distributed between the two study sites. With a small portable generator, a technician can prepare the wells for sampling and do the sampling in about two hours per well.

Between sampling, the wells are sealed with a screw-on cap. There is no cross-contamination between wells and the volume of water removed before sampling enhances sample representativeness. Considering the RCRA requirements for quarterly sampling, or more frequent sampling if a ground-water assessment program is mandated (40 CFR part 265, Federal Register, May 19, 1980, vol. 45, no. 98, pp. 33239-33242), the use of submersible pumps becomes increasingly cost-effective with time.

Initial sampling wells at the California site were constructed with threaded joint ABS casing, with PVC bottom plates bonded to the PVC with two-part epoxy cement. ABS pipe was chosen to facilitate rapid completion of the air rotary borings. The common presence of certain organic constituents in the wastes and in the plastic pipe or epoxy cement precluded further use of these materials. Steel casing and screen was used exclusively in subsequent phases of the California study and throughout the work at the Arizona site.

Decontamination of steel casing and stainless steel screens was accomplished by cleaning with steam, acetone and methyl alcohol to remove solvents and oils. The lacquer coating on steel pipe is essentially impossible to remove. Thus, it is recommended that pipe be ordered which is either uncoated or which has been allowed to develop a coat-

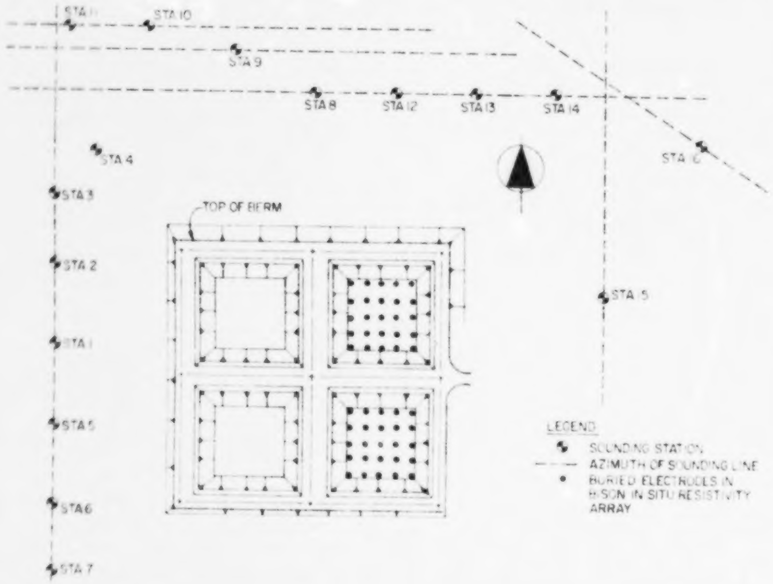


Figure 6. Surface resistivity sounding survey.

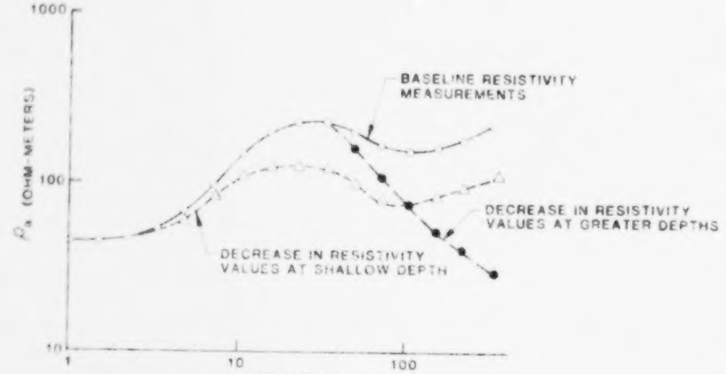


Figure 7. Changes in baseline resistivity measurements due to migration of waste fluids.

ing of surface rust.

Earth Resistivity Soundings

Numerous investigations incorporate electrical resistivity surveys as a means of detecting shallow ground-water contamination (e.g. Stollar and Roux, 1975; Cartwright and McComas, 1968; Warner, 1969). However, not much use has been made of such surveys to establish baseline conditions. At the Arizona site, 16 earth resistivity survey stations were established peripheral to the wastewater pond complex (Figure 6) to define baseline resistivity. Future readings will be taken at these stations. Should earth resistivity significantly decrease, other actions such as test drilling or supplementary readings would be taken to confirm the presence or extent of a leachate problem.

At each station the survey lines were oriented parallel to the pond periphery. The method used was the Schlumberger electrode array which is widely used in electrical prospecting (Dobrin, 1960; Zohdy, 1969). It consists of four electrodes which are systematically extended outward along a straight line from a central point. The (L) spacings used for the outside A and B current electrodes ranged from 1 meter to 215 meters from the central point for a total spread length of 430 meters. The inside or M and N potential electrodes were placed from 0.3 meter to 10 meters. At each spacing the voltage measured was recorded. The apparent potential, ρa, is calculated as $\rho a = K(V/I)$ where K is a constant for each particular electrode spacing, V is the measured voltage recorded on the receiver, and I is the current input level. The current is generally constant for distances less than 100 meters. This is known as the vertical electrical sounding (VES) method, because the increasing spread of the electrodes results in a deeper penetration of the field of current.

The equipment used, a Bison Instruments Inc. Model 2390 signal enhancement earth resistivity system, was the same as that used for the electrode detection system under the waste ponds. The transmitter containing a 12-volt battery has a current output ranging up to 100 milliamperes at frequencies up to 5 Hz. The receiver, with two six-volt batteries, records the voltage up to 1999 millivolts.

The electrical resistivity method of exploration is generally suitable for delineating areas with favorable aquifer conditions when used in conjunction with exploratory borings to provide control. Without correlation with test borings, interpretations of hydrogeologic conditions such as depth, thickness, saturation and physical properties of various layers are unreliable (Stollar and Roux, 1975). Many variable conditions affect the measurement of the various layers, including density, porosity, mineralization, moisture content, formation thickness and the characteristics of overlying and underlying layers. Figure 7 is a schematic showing baseline versus subsequent contaminated conditions as reflected by earth resistivity measurement. Note that the curves are depth sensitive and can indicate which layer(s) are likely to be affected. For monitoring purposes, deviations from baseline conditions are most relevant and detailed interpretation of subsurface conditions is not necessary. Should contamination be detected, however, borings to establish subsurface control and possibly to provide for water sampling are likely to be necessary. At that point, detailed resistivity interpretation should be possible.

Electrical Leak Detection Systems

Two evaporation ponds at the Arizona site have an electrode detection system (patent applied for) installed beneath the lower liner. The system (in-situ resistivity array) is designed and constructed to measure the apparent resistivities of soils beneath the ponds. As in the use of surface electrical resistivity surveys, decrease in the apparent resistivity indicates the presence of contaminants—in this case, the leakage of conductive fluids through the lower line.

As shown in Figure 8, the system for each pond consists of 25 electrodes placed in a five-by-five square grid pattern with adjacent electrodes 45 feet apart. The grid pattern forms multiple Wenner arrays. The electrodes are galvanized steel rods ⅝-inch in diameter and two feet long. They are connected to a junction box by 16 gauge single conductor shielded, coaxial cable. The cables are shielded to minimize electromagnetic interference. Connections between the electrodes and the cable are covered with grease to minimize corrosion, and then protected by heat-shrink tubing.

Prior to installation, preliminary tests were performed to assess the extent of electromagnetic coupling (and the resulting interference with readings) between cables bunched together in narrow trenches as compared with cables spread out, using trenches with a 3' x 3' cross section and as much cable separation as possible.

There were no significant differences observed in apparent resistivity values measured with bunched cables versus those spread out.

The electrodes and cable were installed horizontally in shallower trenches 18 inches deep and 8 inches wide. After installation, the trenches were backfilled and compacted. Figure 9 shows the arrangement and details of the system. All cables terminate at a junction box placed in a protected housing located near a corner of the pond, a maximum slope distance of 75 feet from the nearest electrode.

Operation of the system consists of directing an electric current to one set of electrodes and measuring the potential drop across adjacent electrodes. The potential drop is a measure of the conductivity of the soil and interstitial water between the electrodes. Surveys of all the possible Wenner arrays in the grid system are performed quickly by connecting an earth resistivity meter to the junction box in any one of 28 configurations (see Figure 8). Any leakage of wastes through the lower liner will be detected by a decrease in resistivity values with respect to the baseline readings. The location and rate of leakage migration can be established based on resistivity changes from one date to the next.

Figure 10 shows average resistivity readings from the electrode system beneath the wastewater ponds in Arizona. The readings were taken prior to installation of the liner and reflect ponding of rainfall in the west central sector of the pond, where resistivity is lowest. The contours simply reflect average soil moisture conditions. The spread in values is rather small and on the

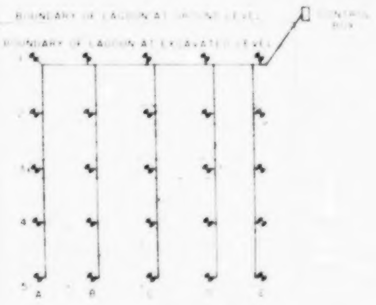

BOUNDARY OF LAGOON AT GROUND LEVEL
CONTROL BOX
BOUNDARY OF LAGOON AT EXCAVATED LEVEL
1
2
3
4
5
A
B
C
D
E

Figure 8. Layout of in-situ resistivity array/electrode detection system.

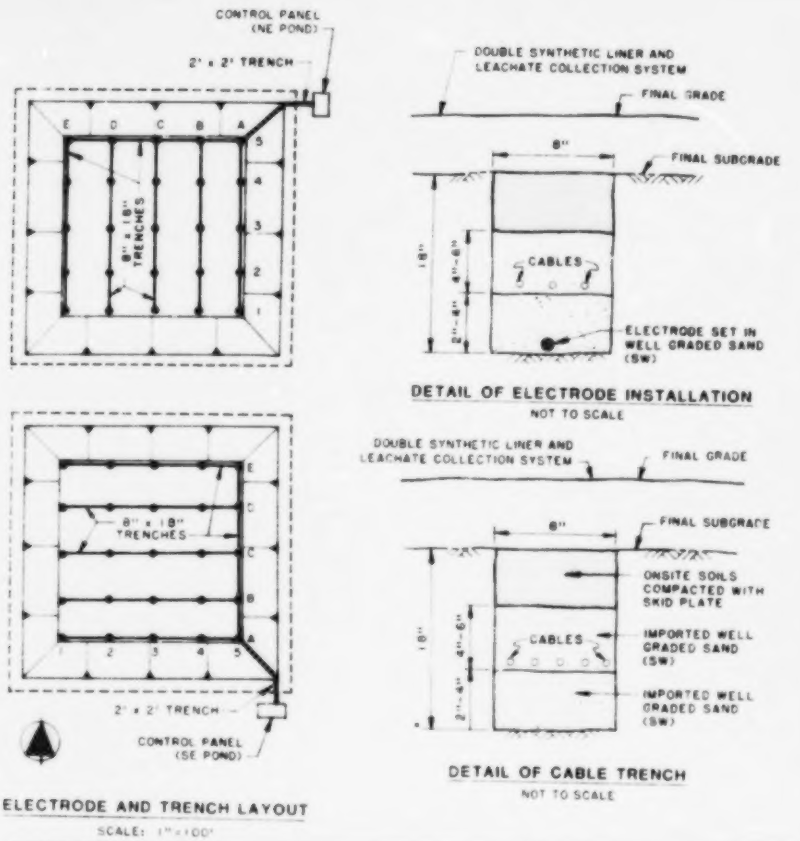

CONTROL PANEL (NE POND)
2' x 2' TRENCH
E D C B A
5 4 3 2 1
8" x 18" TRENCHES
DOUBLE SYNTHETIC LINER AND LEACHATE COLLECTION SYSTEM
FINAL GRADE
8"
FINAL SUBGRADE
18"
4"-6"
2"-4"
CABLES
ELECTRODE SET IN WELL GRADED SAND (SW)
DETAIL OF ELECTRODE INSTALLATION
NOT TO SCALE
ONSITE SOILS COMPACTED WITH SKID PLATE
IMPORTED WELL GRADED SAND (SW)
DETAIL OF CABLE TRENCH
NOT TO SCALE
CONTROL PANEL (SE POND)
ELECTRODE AND TRENCH LAYOUT
SCALE: 1" = 100'

Figure 9. Electrode detection system (patent applied for).

order of 11 ohmmeters. Should a leak develop, the resistivity would be drastically lowered, presumably at a few node points at the onset of leakage.

Water Sampling

Field procedures for sampling wells varied to some degree with the method of construction of the well or piezometer and whether a submersible pump had been permanently installed. For those wells fitted with a pump, sampling was preceded by two to three minutes of pumping. Wells were previously fully evacuated or the water volume in the casing was exchanged five times before sampling. Large-diameter wells without pumps were first evacuated and then sampled using a separate PVC bailer for each installation. The small-diameter wells were sampled using a one-inch outside diameter copper or steel bailer fitted with a foot valve. These wells were sampled after specific conductance values stabilized, an indication that formation water was entering the borehole and casing. In general, conductivity initially rose sharply and then stabilized as formation water displaced the potable water used for drilling.

Between sampling points, the bailers were thoroughly cleaned to avoid cross contamination. This initial procedure was judged to be too time consuming. Subsequent sampling at the California site and all sampling at the Arizona site involved no filtering or sample preservation in the field, minimal reuse of bailers, and more stringent decontamination where needed. The samples were iced immediately after collection and stored in a styrofoam chest for delivery to the laboratory later that day. A preservative (nitric acid) was added to the metal samples on delivery to the laboratory.

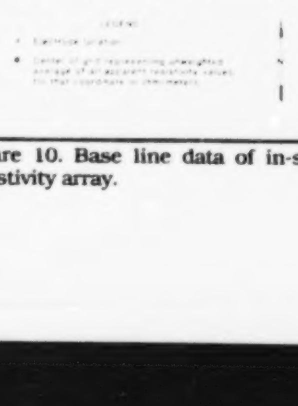


Figure 10. Base line data of in-situ resistivity array.

Characterization of Natural vs. Introduced Organics in Ground Water

Naturally occurring organics in the form of crude oil and gas were present beneath and adjacent to the California site. In addition, the following organics were disposed over an 11-year period:

	Gallons	Tons
Alcohols/Solvents	335,000	—
EDC (ethylene dichloride)	—	1,400
Grease	—	430
Polychlorinated biphenyls (PCB)	—	1
Pesticides	35,000	—
Pesticide containers	—	2,300
Petrochemical waste (including refining waste and petroleum distillates)	270,000	—
	640,000	4,131

Interpretation of the ground-water quality data considered differences in spectra resulting from natural vs. waste-related organics. After generation of gas chromatograph mass spectrometer spectra data, examples of which are shown in Figures 11 and 12, the spectra were analyzed to determine the nature and source of the compounds detected.

Organics found in ground water at the California site originate from one or more of the following: natural crude oil, waste materials, drilling fluids, casing screen and decontamination fluids. The presence of natural crude oil compounds showed up in several deep well water samples as a hydrocarbon envelope (see Figure 11) in the base neutral organics fraction. The hydrocarbon spectra usually consisted of clusters of up to two hundred peaks that easily mask the presence of other waste-related compounds. Most waste-related constituents showed up in the volatile organics fraction and included compounds such as trichloroethylene, trichlorethane, dichloroethane or other halogenated organics.

Compounds considered to be derived from the wastes were primarily methyl benzene, ethyl benzene and toluene. Compounds which were considered to originate with either wastes or well construction included volatiles such as acetone

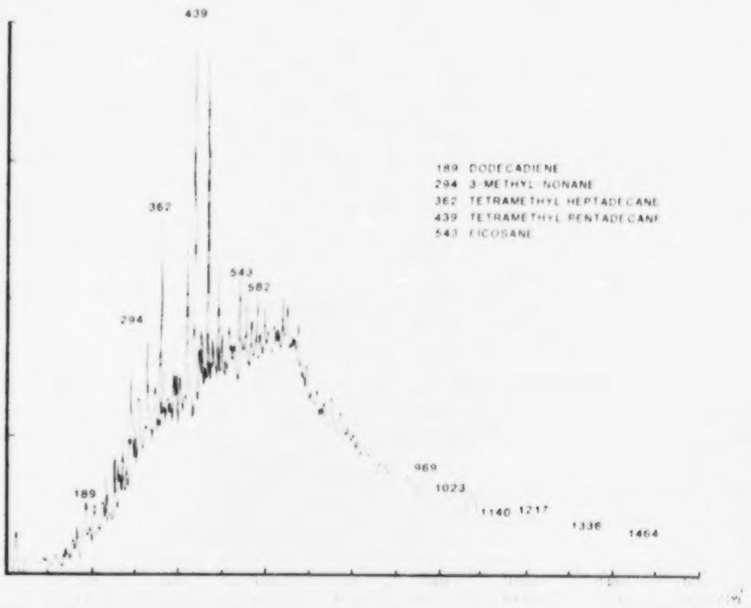


Figure 11. Chromatogram of base neutral organics in a ground-water sample containing natural hydrocarbons.

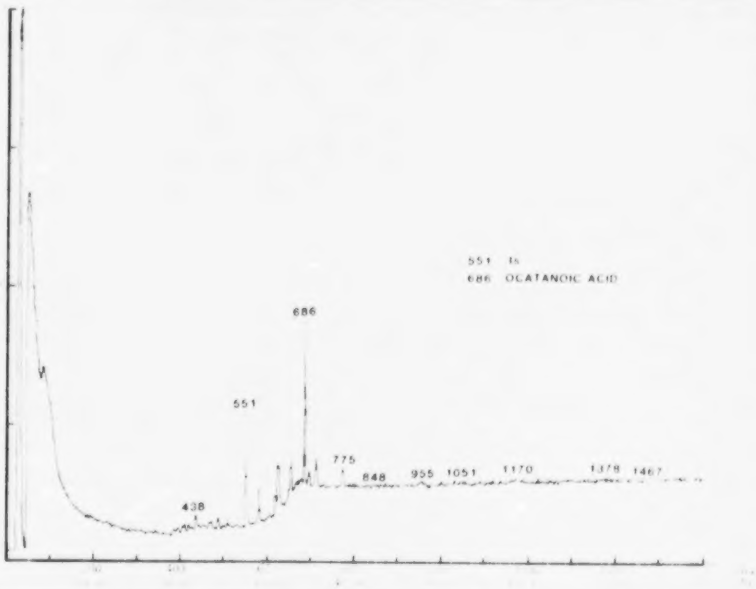


Figure 12. Chromatogram of base neutral organics in a ground-water sample free of natural hydrocarbons.

tetrahydrofuran and methyl ethyl ketone. Some water samples contained elements and compounds which were not consistently found with repeated sampling or contained substances in very low amounts (one to several parts per billion). These results are generally inconclusive and indicate the need for more data to establish a pattern which can be related to ground-water conditions.

It is assumed that ground-water flow in the waste site is laminar and that flow paths and sources, or sinks, are generally constant. Pronounced changes in concentration with time are believed attributable to other than waste or natural factors and may include sampling and decontamination procedures, for example.

The primary natural factor affecting hydrocarbon concentrations would presumably be flushing action associated with seasonal changes in water level, a phenomenon which is noticed more in some wells than in others.

General conclusions reached are that when natural hydrocarbons are abundantly present, they are readily recognizable but they may mask certain waste related constituents. Compounds introduced by the wastes or by the well installation either appear outside the natural hydrocarbon envelope or appear in the absence of such an envelope (Figure 12), as do many organics derived from wastes (e.g. benzene, ethyl benzene, toluene, etc.). Wells yielding generally inconclusive results require additional monitoring with respect to seasonal water level changes and additional sampling.

Summary

Selected ground water monitoring techniques are presented for use in waste disposal facility investigations in arid zone settings of the Southwest. Although local site conditions in terms of geology, hydrology and waste disposal practices largely dictate a given monitoring program and, in turn, the mechanics of instrumenting a site, the following is offered for consideration:

- Completion of wells in poorly consolidated bedrock and in alluvium is greatly facilitated by the use of polymer drilling fluids to stabilize the boring. Air rotary and hollow-stem auger techniques proved disappointing except for very shallow completions.
- Of the PVC, ABS and steel casing and screen materials used, the plastic materials were easiest to install, but presented a possible source of trace organic contaminants in the water samples. Where water sampling for trace organic contaminants is of primary concern, wells should be cased with steel. While galvanized casing was used in wells installed at the sites described, perhaps stainless steel would be even more appropriate, though cost then becomes a significant factor. If plastic is used, blanks should be run to determine the influence of the casing on water quality. Additional quantitative data are needed concerning the full suite of organics and the concentrations that might be expected in water samples from PVC and ABS pipe and associated cements.
- Use of wire-line coring and drive-sampling equipment is recommended considering the relative speed with which representative samples can be obtained. Pitcher barrel samplers proved superior to NX coring in the soft sandstone strata penetrated.
- Penetration of hazardous wastes using driven cylindrical piles was safe and practical for confirming waste thickness and saturation. Water sampling wells could be com

pleted within and below the wastes, but well yields were disappointingly low.

- For purposes of baseline definition and early warning of contaminant release from wastewater ponds, earth resistivity and buried electrode detection systems are excellent methods for inexpensively generating areally distributed, time-series data.
- Although not completely diagnostic, differences in the overall pattern of chromatograms representative of natural hazardous waste-related hydrocarbons are useful in defining contamination.
- In arid zone settings with deep water tables, maximum use should be made of the aforementioned techniques integrated with existing pumping wells in developing least cost monitoring systems to meet the intent of state and RCRA regulations.

Biographical Sketches

The authors are professionals in the firm of Converse Ward Davis Dixon consulting geologists and engineers.

Dr. Robert F. Kaufmann is a principal geologist with 13 years of ground-water experience consisting of university research, the U.S. Environmental Protection Agency and private consulting in the central and western portions of the country. His recent projects have focused on industrial and municipal water supplies, ground-water quality, hazardous waste disposal and the influence of mining and milling on ground water.

T. Alan Gleason is a principal in the firm. He specializes in contamination investigations and compliance evaluations for a variety of industrial clients. Mr. Gleason's experience centers around solid (hazardous) waste management facilities, facility siting investigations and environmental assessments and permitting assistance.

Dr. R. Brian Ellwood, vice president, has specialized in ground-water resource evaluations for over 20 years in both the United States and abroad. Projects in recent years have involved investigations of industrial waste treatment-storage facilities and their interaction with local ground-water regime, compliance evaluations of waste facilities in terms of RCRA criteria and consultations in connection with hazardous waste cleanup programs.

Gerald P. Lindsey is a project geologist with formal training and 18 years of experience in engineering geology. He has been involved with numerous siting investigations including dams, LNG facilities and major commercial buildings.

References

Cartwright, K., and M. R. McCanas. 1968. Geophysical surveys in the vicinity of sanitary landfills in northeastern Illinois: Ground Water, v. 6, no. 5, pp. 23-30.

Dobrin, M. B. 1960. Introduction to geophysical prospecting. McGraw-Hill Book Co., New York, N.Y., 446 pp.

Federal Register. May 19, 1980. 40 CFR Part 265—Interim status standards for owners and operators of hazardous waste treatment, storage and disposal facilities: Subpart F—ground-water monitoring. Federal Register, v. 45, no. 98, pp. 33239-33242.

Scalf, M. R., J. F. McNabb, W. J. Dunlap, R. L. Cosby, and J. S. Fryberger. 1981. Manual of ground-water quality sampling procedures. NWWA/EPA series. National Water Well Association, Worthington, Ohio, 19 pp.

Stollar, R. L., and P. Roux. 1975. Earth resistivity surveys—A method for defining ground-water contamination. Ground Water, v. 13, no. 2, pp. 145-150.

Warner, D. L. 1969. Preliminary field studies using earth resistivity measurements for delineating zones of contaminated ground water. Ground Water, v. 7, no. 1, pp. 9-16.

Zohdy, A. A. R. 1969. The use of Schlumberger and equatorial soundings in ground water investigations near El Paso, Texas. Geophysics, v. 34, no. 5, pp. 713-718.

CAUSE AND EFFECT OF CYCLIC CHANGES IN GROUND-WATER QUALITY

by Wayne A. Pettyjohn

Introduction

It has often been reported that natural ground-water quality is nearly constant at any particular site. Not only do records appear to substantiate this assumption, but logic also would lead to the same conclusion if the aquifer was not submitted to a new stress, such as pumping or the introduction of contaminants. Some generally slight changes of reported concentrations of specific constituents, however, would be expected due to differences in sample collection, storage and analytical technique.

On the other hand, with increasing frequency, investigators are finding that ground-water quality can and indeed may change significantly and rather rapidly, perhaps as much as an order of magnitude and within a few hours, respectively, even though there is no source of man-made contamination.

How can seemingly diverse bits of evidence be used to explain these diametrically opposed concepts? In the first place, deeper or confined aquifers generally are characterized by a nearly constant chemical quality that, at any particular site, reflects the geochemical changes and reactions that occurred as the water migrated through confining layers and aquifers from its recharge area to the point of collection or discharge.

The quality of deeper water can change, however, in response to stresses on the aquifer system. Changes in hydrostatic head brought about by pumping, for example, may cause underflow or leakage of more highly mineralized water from deeper or overlying units into the producing zone. This may be due to fluid migration along the well casing or gravel pack or it may be related to leakage through confining beds, abandoned wells or exploration holes. Another cause is brought about by waste disposal, particularly well injection.

Surficial or shallow aquifers, however, are not well protected from chemical changes brought about by natural events or from the effects of pollution. Surficial aquifers are, in fact, the most susceptible to rapid and sometimes dramatic changes in quality. Several years ago Pettyjohn (1971, 1976) described cyclic fluctuations of ground-water quality and, to some extent, speculated on the causes. It is the intent of this paper to discuss in greater detail the mechanisms that lead to cyclic fluctuations since both the cause and effect can have a significant impact on (1) ground-water quality monitoring and determination of background water quality, (2) transport and fate of organic and inorganic compounds, as well as bacteria and virus and (3) monitoring well design and installation.

The site that Pettyjohn (1976) used to develop the concept of cyclic fluctuation was a particularly well instrumented area in central Ohio. As described in the original report the contaminated area lies on the flood plain of the Olentangy River and is bounded on the east and south by small intermittent streams and on the west by the river (Figure 1). The alluvial deposits, underlain by shale, consist of sand, gravel, silt and clay, that range from 15 to

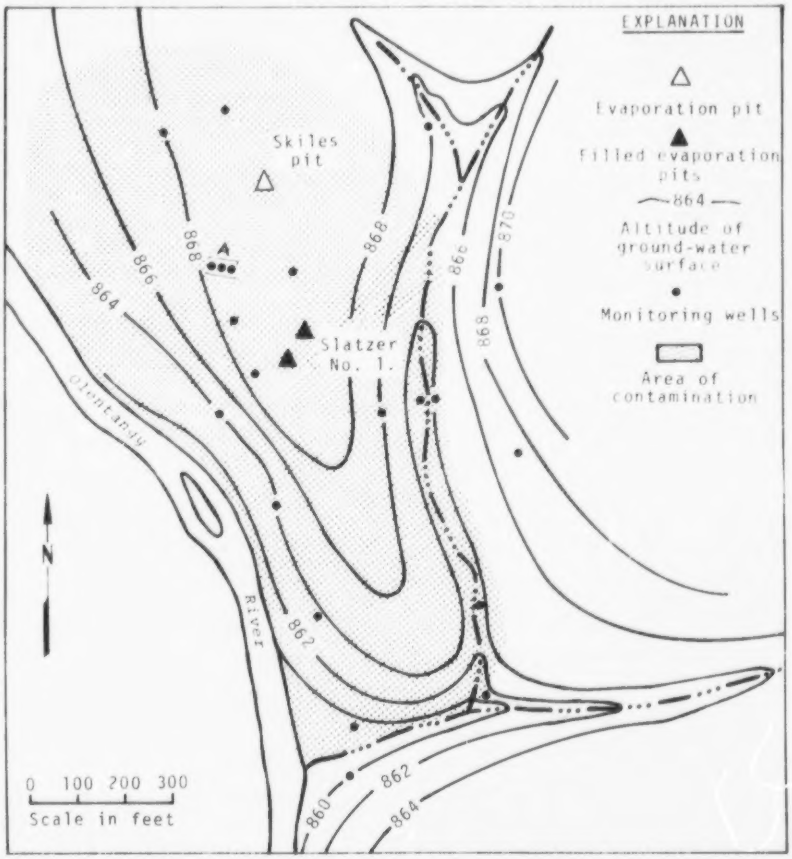


Figure 1. Water-level surface and monitoring wells at the contaminated site

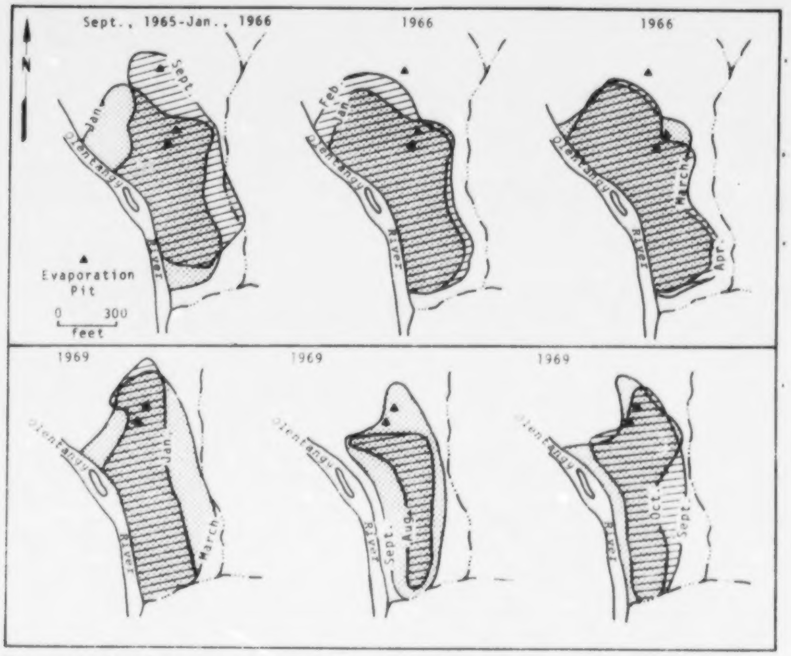


Figure 2. Areal extent of the contaminated area enclosed by the 1,000 mg/l (1965, 1966) and the 500 mg/l isochlors during selected months. Contours based on data from monitoring wells.

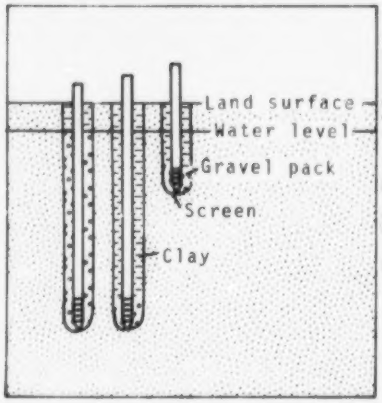


Figure 3 Completion details of three closely spaced monitoring wells.

35 feet in thickness. The water table lies from 1.5 to 5 feet below land surface and fluctuates a maximum of a foot or so throughout the year. Precipitation averages about 38 inches per year.

The site became contaminated during 1964-65 following the excavation of three brine holding ponds. One pit (Skiles) was used from July 1, 1964, to June 30, 1965; about 126 barrels of brine were placed in it. Two other pits (Slatzer), also used about the same time, received 110,000 barrels of brine containing about 35,000 mg/l (milligrams/liter) of chloride.

The brine was spilled on the ground, but largely stored in the holding ponds. Fluids leaked from the ponds and soon contaminated the ground water. When samples were first collected from the 23 monitoring wells in July 1965, the aquifer, at least locally, contained more than 35,000 mg/l of chloride. At this time the ponds were abandoned and the two Slatzer pits were filled with previously excavated materials that had formed surrounding embankments.

An interesting relationship becomes apparent when examining the areal extent of ground-water contamination with time (Figure 2). Note that the area enclosed by the 1,000 mg/l isochlor during 1965-66 changes monthly but that the changes do not necessarily encompass smaller areas. This suggests that a linear flushing rate does not exist. An examination of the 1969 data shows a similar phenomena (Figure 2).

Of particular importance in the monitoring of this site are three adjacent wells, one screened at a depth of 9 feet and another at 23 feet, while a third is gravel packed through much of its length (23 feet) and receives water from the entire aquifer (Figure 3). The locations of these wells are shown in Figure 1 at the position marked A. It is assumed that the first two wells represent the quality that exists at depths of 9 feet and 23 feet respectively and that the gravel-packed well provides a composite sample of the reservoir. It is also assumed that when the composite well has a higher concentration than both the deep and shallow wells, the most highly mineralized water lies between 9 and 23 feet.

The chloride fluctuations in the three wells during 1965, 1966 and 1969 are shown in Figure 4. Notice that at certain times the highest concentrations occur at the shallowest depths, at other times at the greatest depth, and at still others the greatest concentration must lie somewhere in the middle of the aquifer. The only means for accounting for this is intermittent reintroduction of the contaminant, which is puzzling in view of the fact that oil-field activities ceased before any of these samples were collected.

Another technique illustrating the temporal-vertical distribution of chloride in the aquifer is shown in Figures 5 and 6. These illustrations are based on monthly data obtained from the three adjacent wells. Concentrations at depths of 9 and 23 feet were measured; interpretation of the chloride distribution between these points was based on data from

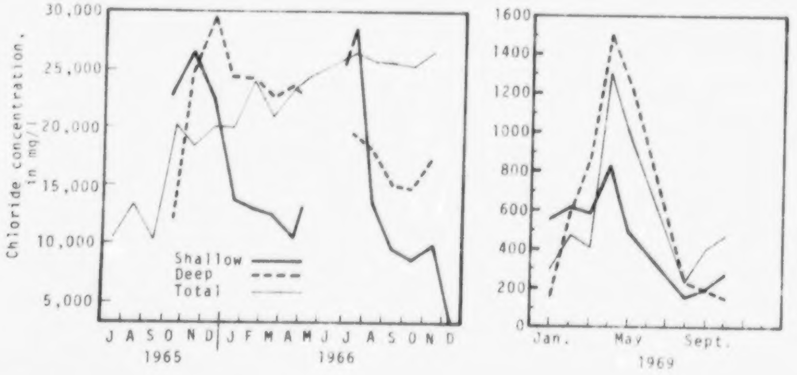


Figure 4. Fluctuation of chloride content in closely spaced wells of different depths during 1965, 1966 and 1969.

the fully penetrating well. In October and November 1965, the highest chloride concentrations were present at the shallowest depth but during December 1965 to April 1966 the highest concentrations were near the bottom of the aquifer (Figure 5). Furthermore, in November and December, the water in the middle of the aquifer was less mineralized than that above or below.

Although greatly reduced, in January 1969 the largest concentration of chloride was again at the shallowest depth, but the situation was reversed during April and May (Figure 6). During February and March the central part of the aquifer was less mineralized than adjacent parts. By August there was only a slight chloride increase with depth, but in September and October the greatest concentrations again appeared in the central part of the aquifer.

The chloride fluctuations that occurred during 1965-66 and 1969 are shown schematically in Figure 7. The October 1965 samples apparently were collected shortly after a recharge event, which leached salt from the ground and formed a highly concentrated mass. This dense, slowly sinking mass (1) was subsequently replaced with less mineralized water. A month later mass 1 had reached and was migrating along the bottom of the aquifer, when another recharge event occurred (2). By December, mass 2 had reached the bottom of the aquifer and was moving towards the river. Recharge events also occurred in January 1966 (3) and in February (4). The February event, however, represented the spring runoff when evapotranspiration was minimum and the soil-moisture content exceeded field capacity over a wide area. This major period of recharge caused a large influx of salt and by March, the aquifer was contaminated throughout a wide area. This mass eventually discharged into the river.

In spite of the fact that brine disposal ceased by mid-1965, the aquifer again was recontaminated several times during 1969 (Figure 7). Following an established pattern, small recharge events took place in January, February and March 1969.

This study indicates that water soluble substances that lie on the hand surface or in the unsaturated zone may be intermittently introduced into a shallow aquifer for many years. The introduction of these contaminants is dependent upon the chemistry of the waste and the soil and on the frequency of the recharge events, which are controlled by evapotransporation, by the rate, duration and intensity of precipitation, and by soil-moisture conditions.

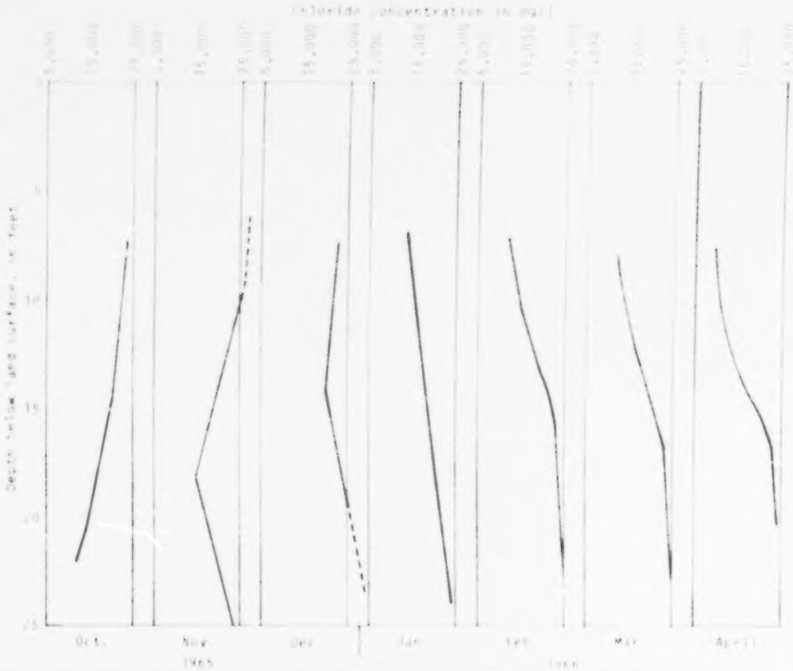


Figure 5. Vertical distribution of chloride at the contaminated site from October 1965 to April 1966.

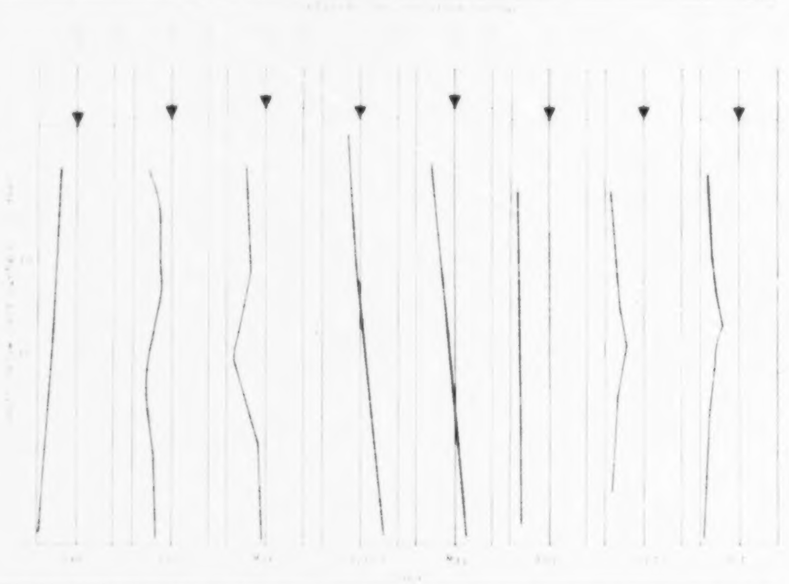


Figure 6. Vertical distribution of chloride at the contaminated site during 1969.

Throughout most of the year in humid and semiarid regions, the quantity of water that infiltrates and the amount of contaminants that are washed into an aquifer are relatively small. On the other hand, during the spring recharge period, great quantities of contaminants may be flushed into the ground over wide areas. Therefore, the major influx of contaminants occurs on an annual basis, although minor recharge events may occur at any time.

Mechanism Leading to Cyclic Fluctuations

In order to account for cyclic fluctuations in ground-water quality it is assumed that (1) the unsaturated zone may store, for long periods, a great mass of water soluble substances and (2) the main paths along which contaminants move through the unsaturated zone to

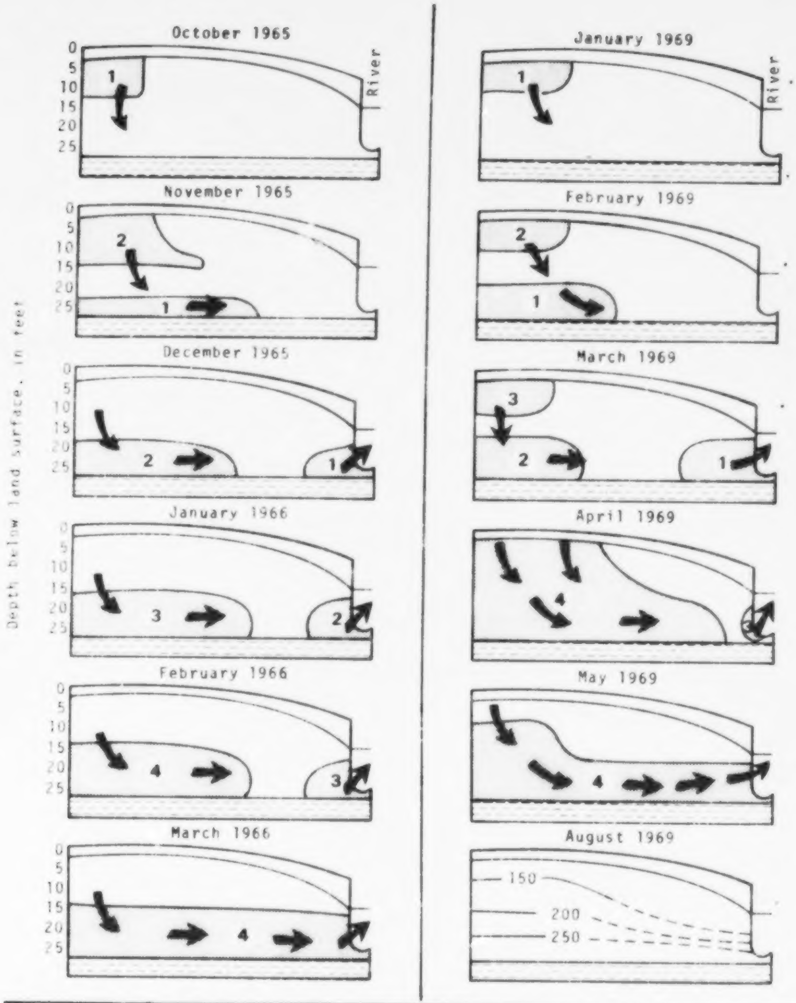


Figure 7. Schematic diagram showing the cyclic movement of masses of contaminated water through the aquifer during selected months in 1965, 1966 and 1969. Stippled areas for 1965-66 represent concentrations in excess of 20,000 mg/1 and for 1969 represent concentrations greater than 500 mg/1.

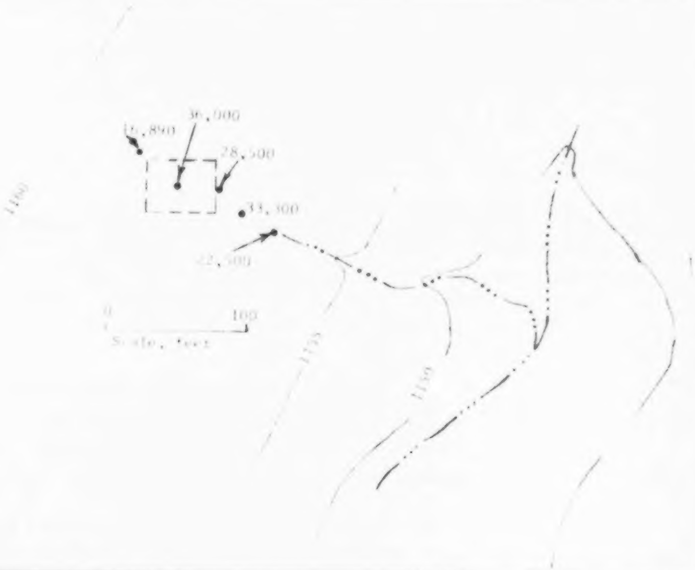


Figure 8. Chloride concentrations in shallow ground water eight years after reclamation of an oil-field brine holding pond.

the water table consist largely of fractures and macropores.

Storage in the Unsaturated Zone

Large concentrations of many water soluble substances are stored within the unsaturated zone. Nitrate storage has been extensively studied. For example, in parts of western Kansas, the top 3 to 4 feet of unplowed prairie soils have as much as 10,000 to 20,000 pounds of nitrogen per acre (Kansas Dept. of Health and Environment, 1981). A particularly interesting case in Texas was described by Kreitler (1975). In addition to natural substances, such as nitrate, many man-related contaminants are stored for years in the unsaturated zone and their presence leads to continual recontamination of aquifers, as was the case adjacent to the Olentangy River in Ohio.

The long-term effects of oil-field brine holding ponds in central Ohio were investigated for several years by geology students at The Ohio State University. One pond was excavated adjacent to a producing well in 1968. Two years later when the well was plugged, the holding pond was filled, graded and seeded. The chloride concentration in the ground a few inches below the water table in the vicinity of the former pond was 36,000 mg/l some 10 years after operation began and eight years after pond reclamation (Figure 8). This particular area is characterized by a thick and very dense glacial till of low primary permeability.

A few miles to the north is an area in which three ponds were constructed. Apparently they had not been used for many months when the area was investigated since the chloride concentration in them ranged from 100 to 3,200 mg/l. In their vicinity 18 holes were hand augered to the water table where chloride concentrations ranged from 100 to 16,900 mg/l (Figure 9). Most of the wells contained less than 2,000 mg/l chloride. In contrast to the previous area, the latter site consists of glacial till within which are several thin layers of sand and gravel. This accounts for the overall lower chloride concentration.

These two sites, coupled with countless others, raises an interesting philosophical question that has never been seriously considered. That is, should waste disposal be permitted only in areas of very low permeability, such as glacial till, to the exclusion of permeable strata?

Disposal in a clay area leads to a highly concentrated leachate that encompasses a relatively small area. The leachate may discharge at springs and seeps or, in case of a landfill, it may be gathered by a collection system for treatment. In these areas there is less likelihood for extensive ground-water pollution, but the leachate is highly concentrated. Disposal in sandy or permeable areas permits ground-water pollution but it also offers the possibility of dilution, the degree depending on permeability, gradient and aquifer thickness. A site on a flood plain may be desirable for disposal if the stream flow is sufficient to dilute the leachate that will discharge into it.

Pathways for Water Movement

It is commonly assumed that ground-water recharge does not or cannot occur until there is a soil-moisture excess, that is until field capacity is exceeded. Increasing evidence clearly points out, however, that the concept of a distinct wetting front or piston-like displacement flow through the main matrix of the soil is unlikely in a great many cases (Thomas and Phillips, 1979, Elrick and French, 1966, Wild, 1972, Ritchie and others 1972, Thomas and others, 1973, 1978, Blake and others, 1973, Eklers, 1975, Quisenberry and Phillips, 1976, 1978 and Shuford and others, 1977). In fact, much of the flow occurs rapidly through macropores and factures. In a study of the Missouri Ozarks, Aley (1977) showed that water entering macropores contributed five times as much recharge as did diffuse flow. Aubertin (1971) found that in sloping forested lands macropores conducted large quantities of water to depths of 30 feet or more. In the majority of cases, at least during parts of each year, ground-water recharge probably is a function both of flow through macropores and fractures as well as displacement flow.

Some macropores and fractures can be exceedingly large. While drilling a test hole in north-central North Dakota with a rotary drilling rig, circulation was completely lost at a depth of about 20 feet. Even after mixing two bags of bentonite and a bag of bran in the drilling fluid, it was still not possible to regain circulation in a fine sand. At this time a curious circumstance was noticed—about 30 feet from the rig a fountain of muddy water was flowing from the trunk of an oak tree that had been cut down years before.

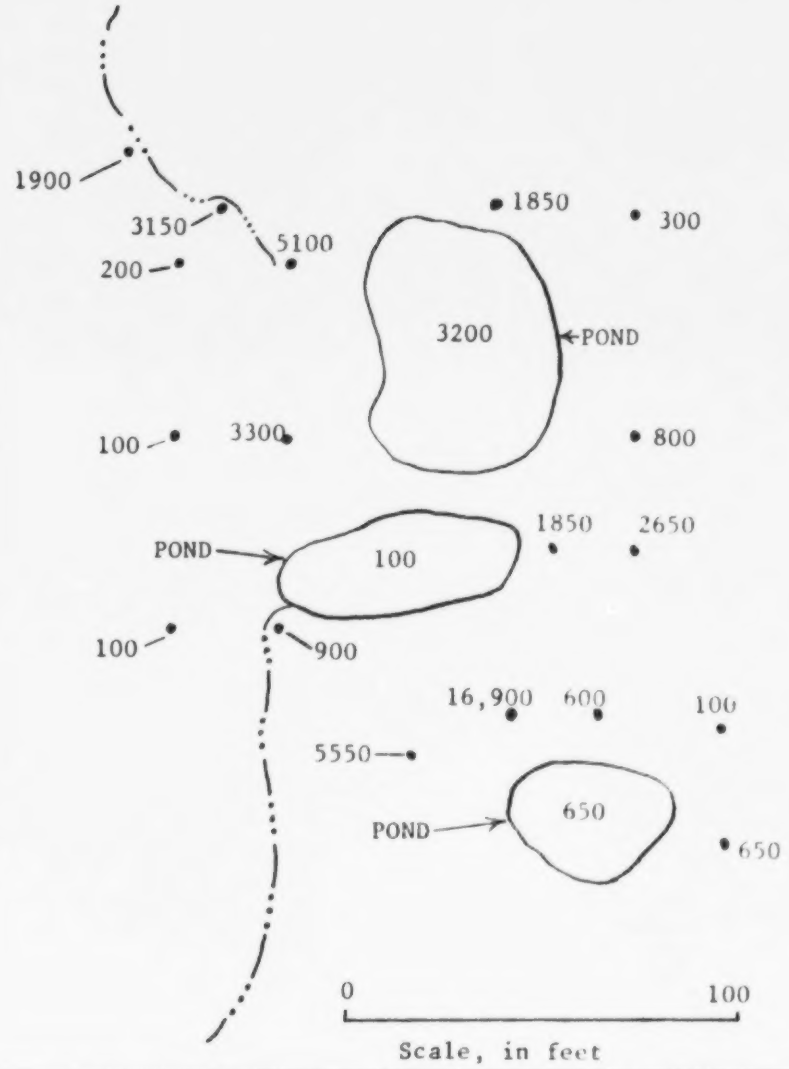
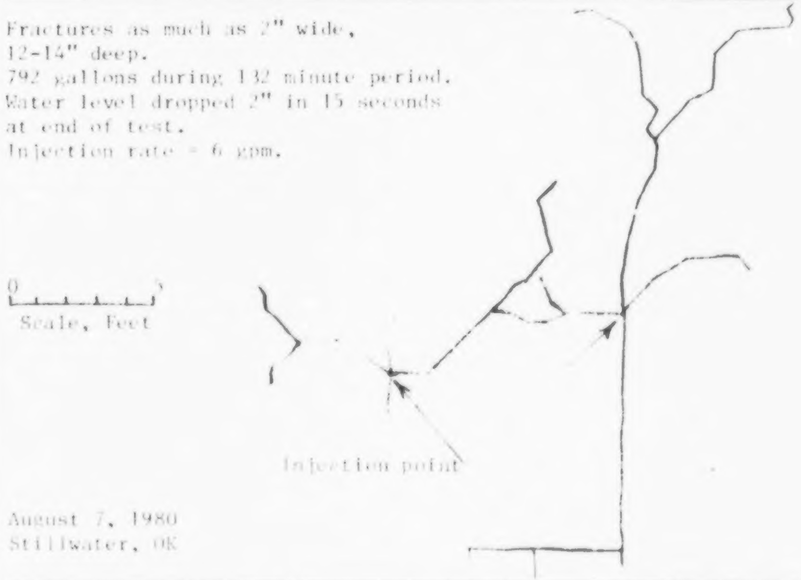


Figure 9 Chloride concentration in shallow ground water in the vicinity of three inactive oil-field brine holding ponds



Fractures as much as 2" wide,
12-14" deep.
792 gallons during 132 minute period.
Water level dropped 2" in 15 seconds
at end of test.
Injection rate = 6 gpm.

August 7, 1980
Stillwater, OK

Figure 10 Sketch map of a desiccation fracture in northeastern Oklahoma

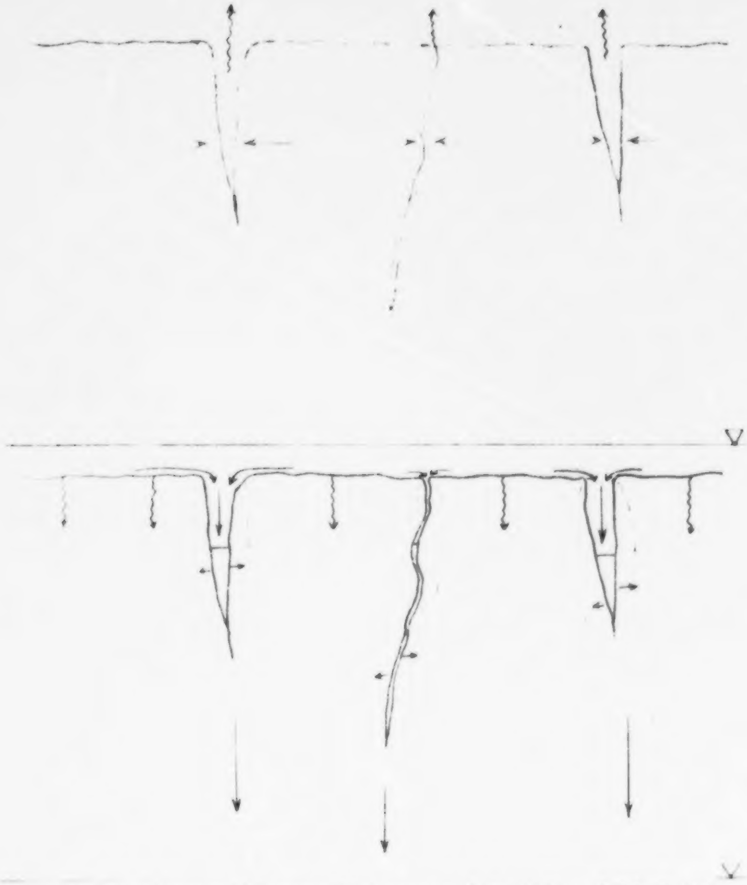


Figure 11. Fractures and macropores accumulate water-soluble linings as a result of evaporation (upper). These linings are dissolved and flushed to the water table during recharge events

Apparently the drill bit had cut through the remains of a large decayed root of the dead tree, and the drilling fluid was following the path of the root back to the surface.

In another case in the same general area, circulation was lost at a depth of about 60 feet while drilling through a thick section of glacial till. After mixing three bags of bentonite with the drilling fluid, circulation was finally regained but not before a spring had formed about 20 feet from the rig. In this case, the drill bit encountered a fracture at some unknown depth that was more permeable than the cutting- and mud-filled annulus of the hole.

The summer of 1980 in northeastern Oklahoma was exceedingly hot and dry. Large, extensive desiccation crack systems were common throughout the finer textured soil areas within the state, including the writer's backyard. One of several such systems was measured on August 7, 1980 (Figure 10). It included a linear distance of about 72 feet, was as much as 2 inches wide and ranged from 12 to 14 inches in depth. A metal probe could be pushed into the crack to a depth of 34 inches.

Two garden hoses, which provided a combined flow of 6 gpm, were placed in two separate locations in the fracture system. Water flowed into the fracture for 132 minutes (792 gallons). At no time did water overflow the fracture. When the water supply was shut off the water level dropped 2 inches in 15 seconds. This primitive test clearly illustrates that fracture systems can collect, store and transmit large volumes of water.

Where extensive desiccation cracks exist, it would be unlikely that much, if any, overland flow could occur except, perhaps, in response to a heavy precipitation event from a convective storm.

Neither macropores nor fractures are commonly as large as those described above. Most may be barely detectable without a close examination. Ritchie and others (1972) suggested that the interfaces between adjacent soil peds also serve as macropores. Moreover, these openings need not extend to land surface in order for flow in them to occur (Quisenberry and Phillips, 1978). Nonetheless, water can flow below the root zone in a matter of minutes. Thomas and Phillips (1979) suggested that this type of flow does not appear to last more than a few minutes "or perhaps, in unusual cases, more than a few hours after cessation of irrigation or rain additions."

Contaminant Transport

The preceding demonstrates that the unsaturated zone can store a large volume of water-soluble substances and that macropores and fractures can serve as highly permeable connecting routes between land surface and the water table. The next consideration is the movement of salts within and from the unsaturated zone to the water table.

When a soil dries, desiccation cracks may form. Moisture on fracture walls may quickly evaporate to develop a strong capillary potential that tends to draw water from the adjacent soil matrix. The soil water may be rather highly mineralized, either with natural substances, such as nitrate, or contaminants that may have been spilled or disposed of on the land surface. Once the soil water reaches fracture or macropore walls and then evaporates, the salts remain as a water-soluble lining.

During a period of rain or irrigation, water may flow into these openings, dissolve the water-soluble lining and rapidly flow downward. This may result in a highly concentrated solution that quickly reaches the water table and substantially degrades ground-water quality. Of course, some of the fluids and salts migrate back into the soil matrix because of the now reversed capillary potential (Figure 11).

Even though there may be a considerable influx of contaminants through macropores and fractures to the water table following a rain, the concentration of solutes in the main soil matrix may change little, if at all. This is clearly indicated in studies by Shuford and others (1977) and again shows the major role of large openings. On the other hand, in the spring, when the soil-moisture content exceeds field capacity, some of the relatively immobile or stagnant soil water may percolate to the water table. A similar widespread recharge period may occur in some places during the fall as a result of decreasing temperature and evapotranspiration, and wet periods that might result in a moisture content in excess of field capacity.

In summary, the preceding suggests the following. During spring, a large quantity of water-soluble substances may be leached from the unsaturated zone over a wide area and cause a significant change in ground-water quality. Although the quantity of leached substances is larger than at any other time during the year, the change may occur more slowly and the resulting concentration in ground water may not be at a maximum because of the diluting effect brought about by the major influx of water over a wide area. A similar but less dramatic change in quality may occur in the fall.

During summer months, ground-water quality changes would be expected to occur more rapidly, perhaps in a matter of hours, because of the large size and abundance of the macropores and fractures. These changes, however, may occur over a relatively small area because of the local nature of convective storms.

Conclusions

One would expect different microbial populations (types and numbers) as well as chemical conditions in macropores and fractures as opposed to the main soil matrix. This may help to explain why some biodegradable organic compounds or those that should be strongly absorbed actually may reach the water table and move with the ground water. This environment cannot be adequately examined by means of column studies.

Some public health investigators have reported (Gerba, 1981, personal communication) that water-borne diseases seem to increase in the spring and fall. This might conform to and be the result of ground-water recharge through macropores and fractures.

In order to detect and evaluate cyclic fluctuations in ground-water quality and determine background concentrations as well, it will be necessary to install monitoring wells that can be used to obtain head measurements and water samples from discreet sections of the aquifer. Moreover, it will be necessary to collect data frequently, perhaps weekly or even daily, until a pattern can be established.

References

Aley, T. 1977. A model for relating land use and ground-water quality in southern Missouri in Dilamarter, R.R. and S.C. Csallany, Hydraulic problems in karst regions: Western Kentucky Univ., pp. 232-332.

Aubertin, G.M. 1971. Nature and extent of macropores in forest soils and their influence on subsurface water movement: USDA, For. Serv. Res. Paper NE 192, 33 pp.

Blake, G., E. Schlichting and U. Zimmerman. 1973. Water recharge in a soil with shrinkage cracks: Soil Sci. Soc. Am. Proc. N.37, pp. 669-672.

Ehlers, W. 1975. Observations on earthworm channels and infiltration on tilled and untilled loess soil. Soil Sci., v. 119, pp. 242-249.

Elrick, D.E. and L.K. French. 1966. Miscible displacement patterns on disturbed and undisturbed soil cores. Soil Sci. Soc. Am. Proc., v. 30, pp. 153-156.

Kansas Department of Health and Environment. 1981. Summary of seminar on nitrates in Kansas. Kansas Dept. of Health and Environment, 30 pp.

Kreitler, C.W. 1975. Determining the source of nitrate in ground water by nitrogen isotope studies. Texas Bureau Econ. Geol., Rept. Inves. 83, 57 pp.

Pettyjohn, W.A. 1971. Water pollution by oil-field brines and related industrial wastes in Ohio. Ohio Jour. Sci., v. 71, no. 5, pp. 257-269.

Pettyjohn, W.A. 1976. Monitoring cyclic fluctuations in ground-water quality. Ground Water, v. 14, no. 6, pp. 472-480.

Quisenberry, V.L. and R.E. Phillips. 1976. Percolation of surface applied water in the field. Soil Sci. Soc. Am. Jour., v. 40, pp. 484-489.

Quisenberry, V.L. and R.E. Phillips. 1978. Displacement of soil water by simulated rainfall. Soil Sci. Am. Jour., v. 2, pp. 675-679.

Ritchie, J.T., D.E. Kissel and E. Burnett. 1972. Water movement in undisturbed swelling clay soil. Soil Sci. Soc. Am. Proc., v. 36, pp. 874-879.

Shuford, J.W., D.D. Fritton and D.E. Baker. 1977. Nitrate-nitrogen and chloride movement through undisturbed field soil. Jour. Environ. Qual., v. 6, pp. 736-739.

Thomas, G.W., R.L. Blevins, R.E. Phillips and M.A. McMahon. 1973. Effect of killed sod mulch on nitrate movement and corn yield. Jour. Agron., v. 65, pp. 736-739.

Thomas, G.W., R.E. Phillips and V.L. Quisenberry. 1978. Characterization of water displacement in soils using simple chromotographic theory. Jour. Soil Sci., v. 29, pp. 32-37.

Thomas, G.W. and R.E. Phillips. 1979. Consequences of water movement in macropores. Jour. Environ. Qual., v. 8, no. 2, pp. 149-152.

Wild, A. 1972. Nitrate leaching under bare fallow at a site in northern Nigeria. Jour. Soil Sci., v. 23, pp. 315-324.

SURFACE GEOPHYSICAL METHODS FOR GROUND WATER MONITORING

Lyle V. A. Sendlein and Hasan Yazicigil

Geophysical methods can play two roles in ground-water monitoring: direct measurement of contaminated water and determination of boundary conditions for flow systems. In any area in which the ground water is to be monitored, the flow system must be totally understood before an effective monitoring system can be established. For this reason, a preliminary geophysical survey is often warranted because of the low cost for each data point as a supplement to an exploratory drilling program. The effectiveness of geophysical methods in direct measurement of any contaminated water is greatly enhanced if a series of background measurements can be made.

It should be made clear that geophysical methods by themselves probably will not provide appropriate means of monitoring ground water in most cases; a drilling program will probably be necessary to supplement geophysical data obtained. In establishing an investigation, it is important that the problem be well defined before holes are drilled and geophysical measurements are made. Where geophysical methods are used, it is important that the geologic or ground-water parameters to be measured by the method are defined.

Direct Measurement of Contaminated Water

Geophysical methods available for practical use can sense differences in either electrical conductivity or thermal activity. Conductivity is a function of the interconnections of the pores, the presence of fluid in the pores, and the conductivity of the fluid. If the conductivity of the fluid changes due to contamination, one should be able to detect this change and thus determine the boundaries and degree of contamination of the fluid. This holds true as long as the geologic materials in which the fluid occurs are homogeneous over the area of measurement. If the pore size or degree of interconnection of the pores change, the electrical conductivity also changes. This physical property is the basis for the use of the electrical resistivity method as applied to stratigraphic problems. For stratigraphic studies, if changes in the properties of the rocks do not change, a resistivity change will not be detected.

As more ions go into solution, the fluid should become more electrically conductive. This increase in conductivity will be directly related to the ion concentration, so that under ideal conditions the resistivity could be related not only to the boundary conditions of the body but also to the degree of contamination. An important statement here is "under ideal conditions." As long as other parameters are constant, i.e., pore size and degree of interconnection of pores, the relationship should be dependent on fluid properties alone. If other properties vary, the electrical properties will also vary and the changes in conductivity are not just a measure of the electrical conductivity of the fluid.

It would be reasonable to assume that the thermal properties of a fluid would also be altered by the concentration of chemical activity found in a waste site. This, however, has not been well documented. To our knowledge, thermal methods have not been widely applied at waste sites.

Geologic Framework Studies

Geophysical methods are designed to measure other physical differences that can assist in defining the flow system. These include stratigraphic boundaries, water-table surface and porosity variation.

For any given site, the relationship of the topographic surface, the bedrock surface, and in some cases the orientation of specific bedrock units relative to a horizontal reference plane must be resolved. If these parameters and the types of geologic material present are known, the flow system can be predicted and a drilling and geophysical program can be designed. The most important element of the flow system relative to the site is the location of the ground-water recharge and discharge zones because these control flow past the site. Many areas have very shallow flow systems and therefore the topographic surface plays an important part in the ground-water flow system (Sendlein and Palmquist, 1975). After all of the background data have been collected, the exploration program should be designed to fill in the data gaps to better define the boundary conditions for the flow system.

All of the boundaries except the water table, will be stratigraphic boundaries of one type or another. For example, bedding planes between rock units may be permeability boundaries if the permeability varies between the units, or porosity boundaries if the porosity varies between the units. The flow may be different between two units; for example, it may be fracture flow in limestone units but intergranular flow in a sandstone adjacent to the limestone.

Stratigraphic Boundaries

Each of the geophysical methods provides different means by which the boundary between two rock bodies can be determined. With seismic and electrical methods, the depth of the interface is determined by a fairly straightforward analysis of the data. However, for magnetic and gravity methods, the boundaries are determined by more indirect methods. Seismic and electrical methods are the most appropriate methods for detailed mapping of selected horizons.

For example, such significant contrasts in seismic properties occur between shale and limestone or unconsolidated material and bedrock of any type that stratigraphic boundaries can be easily determined. The continuity of a boundary or a rock body can be traced because seismic velocity can be used as an identifying parameter of a specific body.

Seismic methods are well adapted to defining certain kinds of boundaries with some accuracy. The seismic refraction method has been very useful as shallow work and will be described in more detail. The reflection method has been applied principally to deep work, but recently high frequency seismic reflection has been applied to shallow problems.

Electrical methods are also well suited for defining boundaries. The electrical resistivity method is sensitive to both fluid content and minerals in the rock. For most sedimentary rock situations, the fluid content variations are the most important variables. In igneous and metamorphic terrains, the conducting minerals in the rock are important and may interfere with ground-water studies. The electrical resistivity method is adapted to define not only vertical differences obtained in horizontal strata but is able to define facies changes or irregular boundaries produced by bedrock valleys filled with unconsolidated material. These contrasts in electrical properties can be mapped in a qualitative manner using apparent resistivity measurements. Under some conditions, the depth and true resistivity can be determined.

Gravity and magnetic data are not well suited for these types of measurements although gravity has been used in buried valley settings, and magnetic and electro-magnetic methods have been used to locate metal objects in waste disposal sites.

Water-Table Depth Determination

Of the geophysical methods available, only the electrical and seismic methods are able to determine the depth to the water table. The contrast in both electrical conductivity and density between wet and dry materials varies significantly enough for both methods to be effective. Gravity and magnetic methods are not suited to determining depth to the water table.

Porosity Variation

Seismic methods can be used to estimate porosity through velocity analysis. The values obtained are very general and can only be used in a qualitative manner. By using velocity to determine rock type, one can also qualitatively arrive at porosity characteristics of the material. In general, higher velocity translates to lower porosity.

Porosity has been estimated for some time by electrical resistivity measurements. The concept has been applied to bore hole logging but could be applied on a very qualitative basis to regional problems.

The Geophysical Methods

No attempt will be made here to describe each method because each has received adequate treatment in the literature. A good summary document is "Techniques of Water Resources Investigations of the United States Geological Survey," Chapter D-1, Application of Surface Geophysics to Ground Water Investigations, by A. A. R. Zohdy, G. P. Eaton and D. R. Mabey published in 1974 by the USGS. The document describes the physics, interpretation techniques and problems of each method, as well as case histories of applications. Other references are included at the end of this article for both general and specific information.

A problem that has plagued the application of geophysics to geological problems has been the fact that the earth system does not always respond as the ideal conductor of accoustic or electrical waves, thus, the data garnered from geophysical studies requires extensive reduction and interpretation. For this reason, some refer to geophysics as an art rather than a science.

Some surveys can be conducted by individuals with little knowledge of the method, whereas other surveys require complex equipment and skilled personnel. It is tempting to use the inexpensive equipment and "cookbook" manual to conduct a survey, however, in the long run this is probably more expensive than using an experienced organization.

Seismic Methods

Seismic methods utilized by geophysicists employ the use of reflected and refracted waves. The seismic reflection method will not be discussed because it has not been applied to shallow ground water problems. However, a new application employing reflection of high frequency signals (very short wavelengths) may have application in shallow work.

A simple layered model is assumed and equations derived to utilize measurements made at the surface. An energy source may be provided by dropping a weight or striking a hammer on a metal plate placed on the ground, or by igniting a cap with a booster or a dynamite charge. The energy moves through the earth and is sensed by a detector (geophone). The geophone sends the signal (now in electrical form) back to an instrument which amplifies, filters and records the signal in synchronization with a time system. This system records the true instant of receipt of the energy impulse. Be

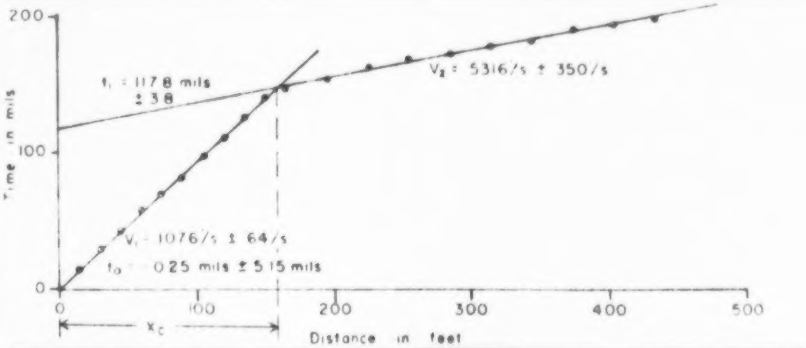


Figure 1. Time-distance graph, two layer case (from Staub, 1969)

cause the distance between energy source and detectors is known, a time-distance plot can be constructed (Figure 1). If the velocity of each layer increases with depth, each slope on the plot represents the reciprocal of the velocity of each layer. In this manner, the velocity of each horizon is known as soon as the plot is constructed. From this plot and from the geometry of the layer shown in Figure 1, the depth to each horizon can be computed by:

$$Z = \frac{V_1 V_2}{2\sqrt{V_2^2 - V_1^2}}\left(t - \frac{x}{V_2}\right)$$

where Z = depth to the interface between horizons
V_1 = velocity through disturbed upper layer
V_2 = velocity through second layer
t = transmission time
x = distance between shot point and geophone

As long as the simple layered model is adhered to by natural earth conditions, this equation holds. But there are some problems that can cause the time-distance plot to become complicated. Figure 2 illustrates ten (10) common cases where the simple layered model does not hold.

Multilayered Case

In Figure 2A, the common multilayered case is illustrated. Each depth must be calculated using the velocity obtained from the time-distance plot. Chances for error increase because each slope of the time-distance plot is a statistical estimate and, therefore, errors are compounded when each deeper layer is calculated. Simple computer programs can be utilized to compute the depth to each horizon and to give the confidence interval (see, for example, Staub, 1969). The basic assumption for multilayers is that each layer has a higher velocity with depth and that the thickness and velocity contrasts are such that each layer produces a line segment on the time-distance plot.

Continuous Velocity Increase with Depth

Figure 2B illustrates the effect of continuous velocity increase with depth without velocity interfaces. This is common in unconsolidated soils and semiconsolidated sedimentary rocks. This type of time-distance plot is not common, but Dobrin (1960) and others have proposed an analytical solution to this condition when it is found.

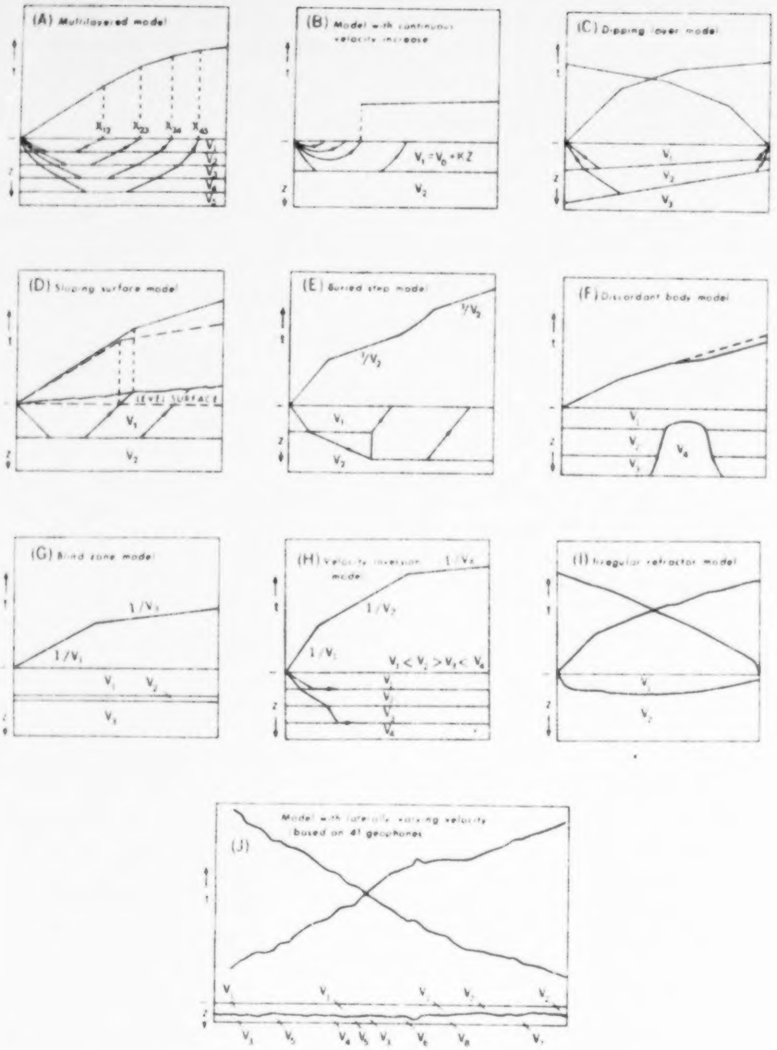

(A) Multilayered model
X_{12} X_{23} X_{34} X_{45}
(B) Model with continuous velocity increase
$V_1 = V_0 + KZ$
(C) Dipping layer model
(D) Sloping surface model
LEVEL SURFACE
(E) Buried step model
(F) Discordant body model
(G) Blind zone model
(H) Velocity inversion model
$V_1 < V_2 > V_3 < V_4$
(I) Irregular refractor model
(J) Model with laterally varying velocity (based on 41 geophones)
t
z

Figure 2. Time-distance graphs for idealized nonhomogeneous geologic models (from Zohdy et al., 1974).

Dipping Layers

Figure 2C illustrates the dipping layer condition. In this case, velocities V_2 and V_3 on the time-distance plot are not the velocities of the second and third layers. To obtain the true depth to the refractors in this situation, one must reverse-shoot the line as shown in the time-distance plot. This means the geophones must be left in the same position and an energy source must be placed first at one end of the line and recorded and then at the other end of the line and recorded. With these data, one can compute the true depth. Reverse shooting should be a common practice in refraction seismic surveying so that this condition can be detected. This is a common occurrence in geologic settings.

Sloping Ground Surface

Figure 2D illustrates the effect of an irregular topographic surface (i.e., a sloping surface). This can be ameliorated if geophone elevations are known by correcting all the geophones to a horizontal datum plane which passes through the elevation of the energy source. Because V_1 velocities are involved, the distance above or below the reference plane can be determined. The time required for the wave to travel over that distance can be computed and either added to or subtracted from the data plotted on the time-distance plot. Another correction often made accounts for a thin surface layer that has a low velocity. This is generally not picked up on the time-distance plot, because the geophones are generally not close enough to the energy source. A preliminary survey with close spacings of the geophones can detect this velocity layer which can then be added to each time-distance plot.

Buried Step and Discordant Bodies

Figures 2E and 2F represent special cases that are not common but can occur. Note that the time-distance plot has a nonlinear segment. When this is observed on a time-distance plot, one has to investigate the reason for such occurrence.

Blind Zone

Figure 2G illustrates a very common cause for error in seismic refraction data. This problem can be caused by the relationship between the thickness of a layer overlying a much higher velocity layer. If the velocity contrast between V_2 and V_3 is high, then the thickness of the layer, too, must be large enough to allow the refracted ray from layer 2 to get enough data points on the time-distance plot before the refracted ray from layer 3 arrives at the surface. This is a common problem in limestone sequences because of the extremely high velocity of limestone. The thinner the layer above the higher-velocity layer, the lower the velocity contrast has to be for this to occur.

Velocity Inversion

Figure 2H illustrates the violation of the basic assumption that velocity increases with depth. When a lower velocity layer is present beneath a higher velocity layer, total refraction does not occur and the energy from the interface does not refract back to a surface. This means that a line segment on the time-distance plot is not present for that layer and the time required for travel through the low velocity layer is added to the next refracting layer. Thus, the next line segment that occurs on the time-distance plot does not represent the true velocity of the succeeding layer. This will cause problems in the calculations resulting in erroneous depth to the next horizon.

Figures 2I and 2J represent the real world. As the refractors become deep, the geophones must be spread further apart. Thus, fewer points define the line segments of the time-distance plot. The time-distance plots then become more difficult to interpret; only experience can assist the interpreter.

The reliability of the depth determined by seismic refraction measurements decreases with depth and increases with the experience of the operator. The slope of the linear segments of the time-distance plot are the primary data points. The number of points on each segment and the corrections made to the points are critical to the reliability of the data. In all cases, the data should be presented in statistical terms (i.e., a given depth plus or minus, a specific confidence interval) to remind the user that the depth determined is approximate.

Application of Seismic Refraction Measurements to Hydrogeologic Problems

There are several applications of the seismic refraction method very appropriate to hydrogeologic investigations. They include bedrock surface mapping, water-table depth determination, lateral facies change identification in aquifers, and mapping of stratigraphic information.

Bedrock Surface Mapping

The interface between the bedrock and overlying unconsolidated sediments in most cases will be a permeability boundary. By mapping this surface, an important boundary of the flow system can thus be determined. Buried valley aquifers in Iowa have been mapped in detail through the use of seismic refraction; two examples are presented here to illustrate this point. The Nishnabotna River Basin of southwestern Iowa occupies a surface area of approximately 10,000 square kilometers (4,000 square miles) and contains a portion of a major buried bedrock valley system beneath it. The bedrock surface has been partially mapped using bore-hole data. The seismic refraction method was employed where insufficient bore-hole data would not allow detailing of the major buried valley underlying the basin divides (Stone, 1971). This study included the development of flow maps and identification of places where ground water was entering or leaving the basin via the buried-valley system.

A second study determined the location of bedrock-valley aquifers which supply water for the city of Ames, Iowa. The study included an analysis of the impact of a proposed dam on the Skunk River which roughly parallels the buried bedrock-valley system, a detailed analysis of the well field supplying the city of Ames, and an analysis of the impact of hydrocarbon pollution sources on the aquifer system.

The determination of the precise boundaries of the aquifer was paramount to these analyses. Seismic refraction data played an important role in this aspect of the study.

Through seismic profiling, the major axis of the buried valley was determined and, in the vicinity of the Ames well field, the bedrock was detailed so that the Pleistocene stratigraphy could be determined. In the area studied, the bedrock surface is composed of Mississippian limestone and Pennsylvanian shales. The high velocity of the limestone (12,000 to 14,000 feet per second) and lower velocity of the shales (7,000 to 8,000 feet per second) provided an easy means of determining the nature of the bedrock material. Velocity contrasts between alternating layers of sand and gravel and glacial till overlying bedrock likewise made the determination of unconsolidated material stratigraphy an easy task. The information obtained from seismic profiling thus allowed for the construction of detailed aquifer maps.

Water-Table Depth Determination

The contrast between the velocity through saturated and unsaturated sediments provides a velocity boundary generally detected by the seismic refraction method. The low velocity layer represents the unsaturated zone, while the higher velocity layer represents the saturated zone. In some cases, the saturated zone of the sediments above the bedrock is very thin and creates a blind zone which is not detectable with seismic refraction; this is a possible source of error in determining the true depth to the water table.

Lateral Facies in Aquifers

In many studies, enough seismic measurements are collected so that velocity maps of the first layer can be constructed. Any velocity changes encountered can be roughly related to density or porosity changes and, if bore-hole data is available, one can correlate the velocity with variations in aquifer material and a facies map can be constructed. Eaton and Watkin (1967) published such a map as part of a study in Ohio. They were able to map the distribution of compressional wave velocities in unsaturated outwash sand and gravel in the valley of the ancestral Miami River in southwestern Ohio. The variations in velocity represented lithofacies variations in the upper 100 to 300 feet of the valley fill. This information, correlated with productivity data, could be used to determine optimum locations for wells. It should be pointed out that for these kinds of maps to be constructed, the data must be collected as a series of profiles located close enough together to represent a grid pattern.

Stratigraphic Information

Because the nature of the geologic bodies present at a site controls the occurrence of ground water, determining the stratigraphy of the area is important. While the importance of lateral variation on facies changes has been discussed, it is important to comment on the application of the seismic refraction method to other stratigraphic data that can be obtained.

A seismic profiling technique was used by Staub (1969) to map a high velocity limestone unit in the Skunk River Valley. This bedrock layer was overlain by other limestones and shales and Pleistocene sediments. However, it could be identified on the time-distance plot and mapped as a marked horizon. This study detailed a major fault system in the region which was associated with the buried valley of the Skunk River. The high velocity limestone (16,000 feet per second) produced an excellent marker horizon for which seismic mapping of this horizon allowed a structure map to be constructed for the region. With this information and associated bore-hole data, other bedrock units above the marker horizon could be mapped utilizing the bedrock surface map to produce a bedrock geology map of the region.

Arnow and Mattick (1968) present another type of stratigraphic application illustrating how stratigraphic units could be delineated by a seismic profile. In this case, a semiconsolidated unit was discovered beneath an unconsolidated Pleistocene cover. Here again, velocity differences were used to identify a different stratigraphic unit. The feature is of large size, measured in excess of 10 kilometers.

In some cases, stratigraphic horizons of very subtle velocity contrasts can be defined. Johnson (1954) was able to delineate the boundary between two glacial till bodies in Illinois using seismic refraction data. This was possible because the surface was relatively uniform and borehole data was available.

Discussion

The seismic refraction method is a powerful tool in hydrogeology, but it is not a direct measuring tool for contaminated water. The boundaries measured are bedrock, water table and some bedrock stratigraphic surfaces. When used with velocity data (which can be characteristic for given rock units and unconsolidated material) and surface topography information, the best monitoring program to develop for a given site can be designed.

Electrical Resistivity

Among the electrical methods, the resistivity method is the most widely used for ground-water studies. This discussion will be limited to the resistivity method.

The electrical resistivity, or inverse conductivity of a material is defined as being numerically equal to the resistance of a specimen of the material of unit dimensions. The unit of resistivity in the metric system is the ohm-meter (ohm-m).

The range of resistivities among rocks is enormous; it varies from less than one to several thousand ohm-m. In most rocks, electricity is conducted electrolytically by the interstitial fluid, and resistivity is controlled more by porosity, water content and water quality than by the resistivities of the rock matrix. Saturated rocks have lower resistivities than unsaturated and dry rocks. The higher the porosity of the saturated rock, the lower its resistivity. Also, the higher the salinity of the saturating fluids, the lower the resistivity. Presence of clays, argillaceous materials, and conductive materials also reduce the rock's resistivity.

Description of the Method

In an electrical resistivity survey, a direct current or a very low frequency alternating current of intensity I is passed into the ground through a pair of current electrodes (A and B) and the resulting potential drop ΔV is measured across a pair of potential electrodes (M and N) (Figure 1). The resistivity is given by the following formula:

$$\rho = K \frac{\Delta V}{I}$$

where K is a geometric factor depending only upon the relative position of the four electrodes.

Among the several electrode array configurations, the Wenner and Schlumberger configurations are the most widely used in electrical resistivity. In the Wenner method, four electrodes are placed along a straight line so that AM = MN = NB = a, or AB/MN = 3. For this configuration, $K = 2\Pi a$. In the Schlumberger method, the four electrodes are placed along a line in the same order, AMNB, as in the Wenner array, but with $AB \geq 5MN$. For this configuration

$$K = \Pi \frac{AM \bullet AN}{MN}$$

The other arrays (i.e., Lee-Partitioning and Dipole-dipole) are less frequently used. For example, the dipole-dipole array is only used for very deep investigations, when for practical reasons one wants to avoid a very long line AB (Brown et al., 1977).

The resistivity obtained corresponds to the true resistivity only if the geologic medium is homogeneous and isotropic. However, if the medium is nonhomogeneous and anisotropic, which is obviously the usual case, the measured resistivity is an apparent resistivity, the value of which is a function of several variables: the electrode spacing, the geometry of the electrode array, and the true resistivities and other characteristics of subsurface materials such as layer thicknesses, angles of dip and anisotropic properties.

The study of the variations in apparent resistivity is the basis of the resistivity methods. Two techniques, i.e., electrical sounding and horizontal profiling, are employed depending upon whether vertical or horizontal information is desired.

Electrical sounding is used to investigate depth and horizontal profiling is used to detect lateral variations in resistivity. However, the results of electrical sounding and of horizontal profiling are often affected by both vertical and horizontal variations in the electrical properties of the subsurface materials.

The basis for making an electrical sounding, irrespective of the electrode array used, is that the further away from a current source the measurement of the potential difference is made, the deeper the probing will be. Hence, deeper and deeper beds are penetrated as the electrode separation is increased. The results of electrical sounding are usually displayed on log-log paper by plotting the apparent resistivity against a constant fraction of the length AB. The quantitative interpretation of the electrical soundings are then made by matching the field graphs with master curves. For quantitative interpretation of the electrical sounding data, the reader is directed to several excellent references (Zohdy, et al., 1974; Zohdy, 1974; Zohdy and Bisdorf, 1975; Flathe, 1967; Van Nostrand and Cook, 1966; Kunetz, 1966 and Al'pin et al., 1966).

In horizontal profiling, a fixed electrode spacing is chosen and the whole electrode array is moved along a profile after each measurement is made. The results are presented as apparent resistivity profiles or apparent resistivity maps. The interpretation of horizontal profiling data is generally qualitative. They are particularly useful in locating buried stream channels, veins, faults, dikes and in determining the areal extent of a contamination plume.

Application of the Resistivity Method to Hydrogeologic Problems

In ground-water monitoring studies, the resistivity method can furnish information which might be unattainable by other geophysical methods. For example, electrical methods are unique in furnishing information concerning the depth of the fresh-water salt-water interface and the areal extent of a polluted ground-water zone, whereas neither seismic, gravity, magnetic nor thermal methods can supply such information. Cited here are selected examples of electrical resistivity studies drawn from the more readily available literature. For a more comprehensive review of the application of the resistivity method, particularly with respect to ground water, the reader is referred to several excellent books which cover the material much more thoroughly than is possible with an article. These include a review of resistivity methods with applications to ground water by Zohdy, et al., 1974; a discussion on the limitations of the resistivity method by Van Dam, 1976 and Klefstad et al., 1975 and the cost of the resistivity method as a monitoring tool for ground-water contamination studies by Everett, 1980.

Mapping Buried Stream Channels

There are several examples in the literature (Zohdy, 1964 and 1965; Page, 1968; Smith, 1974; and Poole and Heigold, 1981; Lennox and Carlson, 1967) where courses of buried channels were accurately mapped by using electrical resistivity methods. In general, horizontal profiling is used to furnish information on the presence or absence of buried stream channels and electrical soundings for the determination of depth preceding and following the horizontal profiling survey.

Mapping Clay Layers

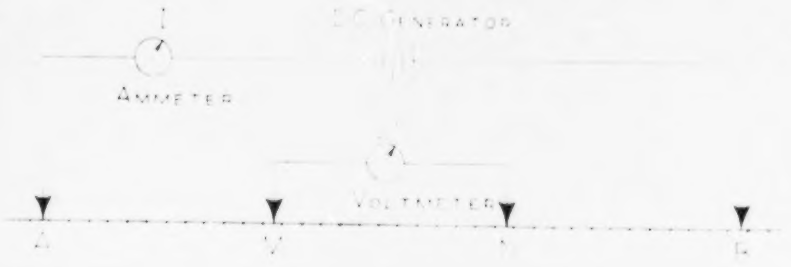


Figure 3. Quadripole AMNB configuration for resistivity measurements

Determination of the presence or absence of clay layers, their thickness and depth is valuable information for waste-disposal and ground-water contamination studies. In most cases, clay layers act as the lower boundary of the contaminant enclave. Electrical resistivity methods can be used to map clay layers because of their relatively low resistivity. Zohdy and others (1974) have successfully used electrical sounding techniques to map a clay layer which separates two aquifers near Bowie, Arizona.

Mapping the Water Table

The determination of the position of the water table by the electrical resistivity method is based on the fact that the saturated materials will have lower resistivity than the unsaturated materials. However, only under favorable conditions in homogeneous deposits can one approximately determine the depth to the water table. In fact, one finds the top of the capillary fringe. This fact should be considered in conducting studies in fine-grained deposits characterized by a large capillary fringe (Ogilvy, 1967). Wherever the water table is overlain and underlain by several layers of different resistivities, its detection on a sounding curve may be virtually impossible.

Determining Aquifer Parameters

Recently there have been quite a few studies on the determination of the porosity and hydraulic conductivity of an aquifer by electrical resistivity measurements (Griffiths, 1976; Heigold et al., 1979; Kelly, 1977; Kosinski and Kelly, 1981 and Keller, 1967). Estimation of these parameters is based on semi-empirical relations developed between resistivity and hydraulic properties of aquifers obtained from pumping tests. The starting point is the formation factor, F, defined by Archie (1942) as:

$$F = \frac{\rho}{\rho w}$$

where ρ is bulk resistivity and ρw is water resistivity. According to Archie, the formation factor, F, is related to porosity, ϕ, by:

$$F = a\phi^{-m} \text{ or } \rho = a\rho w\phi^{-m}$$

where a and m are constants whose values are the best fit estimates to a particular group of measurements. These parameters correlate well with the lithology and texture of a rock. The value for the parameter "a" varies from slightly less than 1 in clastic rocks to as large as 3.5 in vesicular or tuffaceous rocks. The parameter "m" varies from values as small as 1.3 in packed sand to values as large as 2.3 in tightly cemented clastic rocks. Keller (1967) lists typical expressions for Archie's law applicable to various types of rocks.

Although a fairly well defined relationship between porosity and resistivity is developed, the relationship between hydraulic conductivity and resistivity is not well defined. Kelly (1977) obtained a direct linear relationship between hydraulic conductivity and resistivity of the water-bearing deposits. Recently Heigold et al. (1980) found a geometric relation in the form of:

$$H = b\rho^{c}$$

where b and c are constants whose values are the best fit estimates to a set of measured data obtained from pumping tests and resistivity surveys. The empirical relations developed so far would be of limited value in areas other than the study area.

Mapping Freshwater-Saltwater Interfaces

Electrical resistivity methods have been successfully used for mapping the freshwater-saltwater interface for many years. Examples of the application of the technique to locate freshwater saltwater interfaces are Sayre and Stephenson in Texas (1937), Swartz in Hawaii (1937), Van Dam and Meulenkamp in Neth erlands (1967), Flathe in Germany and Indonesia (1967), Zodhy in Texas (1969), Zohdy and Jackson in Hawaii (1969) and Van Dam in Netherlands (1976).

Delineation of Ground-Water Contamination

Recently, electrical methods have been applied for delineation and mapping of contaminated ground water. The method is based on the fact that formation resistivity depends on the conductivity of the pore fluid as well as the properties of the porous medium. Under ideal conditions, contrasts in resistivity may be attributed to a plume of contaminated ground water, with a higher than normal conductivity, originating at a contamination source. Under suitable conditions, a combination of vertical electrical soundings and horizontal profilings can be used to rapidly determine the boundaries of this plume of contaminated ground water (Figure 2).

The literature is rich with examples of the application of the resistivity method in ground-water contamination studies. Success has been reported in tracing water from landfills (Cartwright and McComas, 1968; Stollar and Roux, 1975; Kelly, 1976), acid mine drainage (Merkel, 1972), sewage treatment effluent (Fink and Aulenbach, 1974), septic tanks and oil field brine disposal pits (Warner, 1969), industrial process waters (Berk and Yare, 1977) and spent sulfite liquor (Hackbarth, 1971).

Success with the resistivity method in contamination studies depends to a large extent on a good knowledge of subsurface conditions. Conditions favorable for delineation of zones of contamination include sufficient contrast between the conductivity of the contaminated and the natural ground water, a shallow ground-water table, uniform subsurface conditions, and proper electrode spacing (Hackbarth, 1971). Klefstad et al., (1975) have shown that the electrical resistivity method was unsuccessful in delineating the contaminated ground water in their Ames study because the natural scatter produced by large variations in geologic materials obscured the decrease in resistivity due to contamination. Hackbarth (1971) overcame this problem by making resistivity measurements before and periodically after the disposal of the contaminant so that the effects of lateral variation in geologic materials were effectively eliminated.

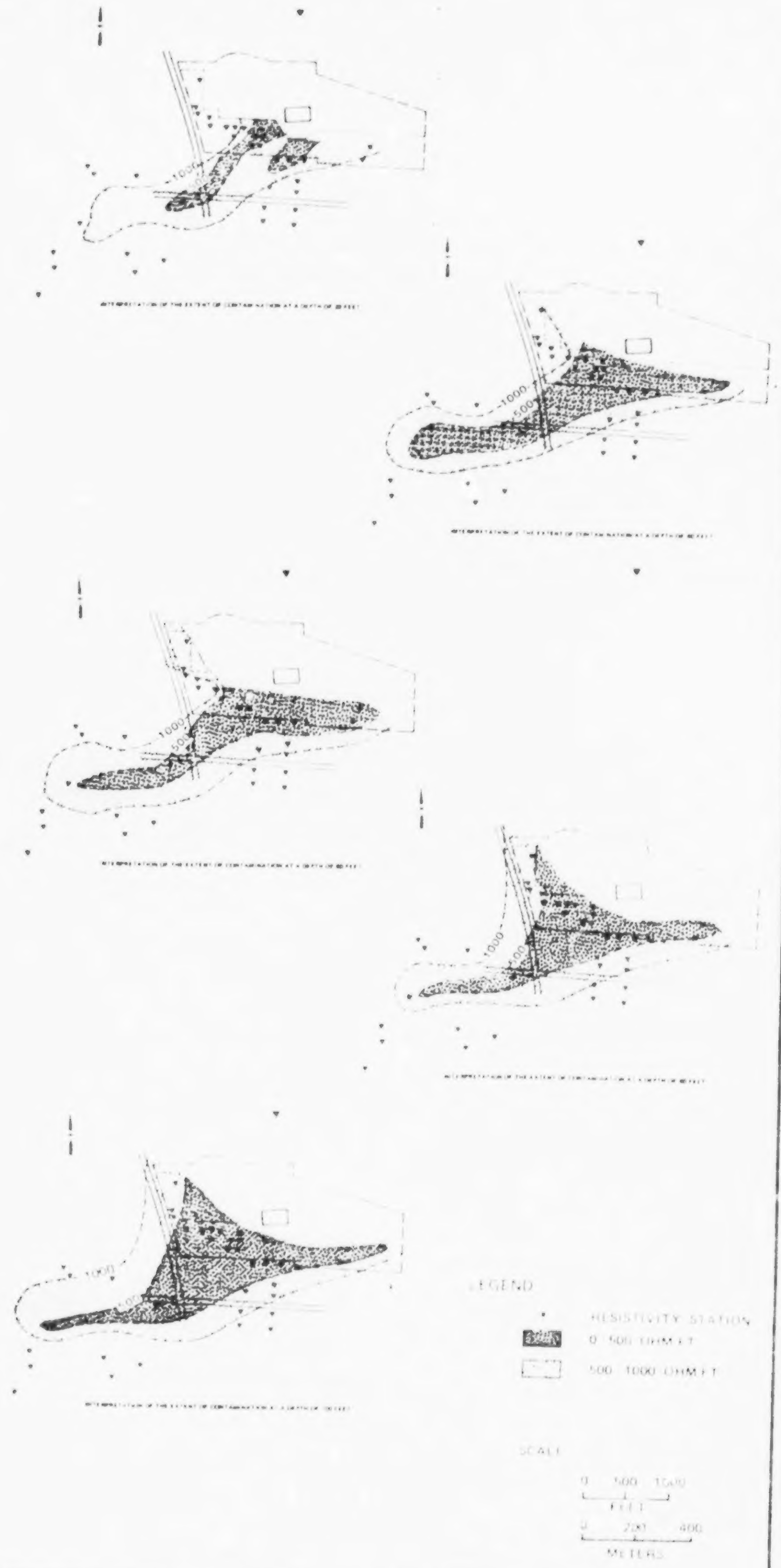


Figure 4. Interpretation of the extent of contamination at various depths below ground surface (from Berk and Yare, 1977)

Delineating Zones of Solution Channels, Major Fracture and Joint Systems in Carbonate Rocks

Resistivity measurements can be useful in finding solution channels, filled sinkholes, major fractures and joint systems, and caves in carbonate rocks or in karst terrains. Contaminated ground water in karst terrains moves rapidly through the interconnected system of solution channels. Therefore, the location of these features is an important step for any ground-water monitoring program in carbonate rocks or in karst terrains. Kirk and Rauch (1977) used the tri-potential resistivity method to locate filled sinkholes, caverns and fracture zones in karst areas of West Virginia and Pennsylvania. Manley and Garton (1977) successfully used the Lee Modification of the Wenner Configuration to find the depth, height and position of cave walls near Bowden, West Virginia.

Discussion

The applications described clearly indicate the usefulness of the resistivity method in ground-water monitoring studies. However, the method has certain limitations. These include poor site access, interference of buried conductors such as pipelines or wires, insufficient contrast between the conductivity of the contaminated and the natural ground water, depth to the top of the contaminated zone, thickness of the zone, and the complexity of the surficial geology (Stollar and Roux, 1975). Furthermore, several other limitations are inherent, such as in the interpretation of the electrical sounding data, and these are discussed in Zohdy and others (1974). The interpretation of a multi-layer sounding curve generally is not unique. This means that a given electrical sounding curve can correspond to a variety of subsurface distributions of layer thicknesses and resistivities. The limitations mentioned above should not be discouraging to the geophysicist or hydrogeologist engaged with ground-water monitoring studies. If these limitations are kept in mind, the resistivity method can provide useful information on the lateral extent of the contamination plume and on the determination of geologic framework at a reduced cost and within a short time period. This reduces the number of drill holes required for monitoring and helps to design an optimal ground-water monitoring program.

Thermal Method

Conventional geophysical methods, i.e., electrical resistivity, seismic, gravity and magnetic measure only some properties of the rock medium in which ground water occurs (electrical conductivity, elasticity, density and magnetic susceptibility) rather than measuring properties of the water itself. A property that may be used in ground water studies is the high specific heat of the water or its resistance to changes in temperature. The specific heat of water is equal to 1.0, whereas that of the majority of dry geologic formations is 0.2-0.3. For this reason, an aquifer located at shallow depth will heat and cool with the seasons much less rapidly than the surrounding impermeable formations. Thus, a shallow aquifer may act as a heat sink or a heat source (depending upon season of the year) that might influence the near-surface soil temperatures to a measurable extent (Lovering and Goode, 1963; Cartwright, 1968a, b, 1970 and 1974). If this anomaly can be detected and distinguished from other factors that affect soil temperatures, thermal methods can effectively be used to delineate shallow ground-water flow systems, buried valley aquifers, recharge and discharge zones, maximum permeable zones, leakage beneath earthen dam embankments and locations of solution channels or narrow fractures containing ground water moving at relatively high rates in karst terrains.

The effect of the movement of ground water on the earth's thermal regime has been discussed by Stallman (1963 and 1965), Bredehoeft and Papadopulos (1965), Parsons (1970), Domenico and Palciauskas (1973) and others. The effect of the variation in thermal regime on soil temperatures and the use of thermal prospecting in ground water studies have been discussed in detail by Cartwright (1968a, b, 1970, 1971 and 1974), Birman (1969), Smith (1974), Schneider (1962), Kremar and Masin (1970), Turner (1977), and others. The following discussion on the use of thermal methods is synthesized from several of the above papers.

Thermal Anomaly Caused by Shallow Aquifers

The effect of a shallow aquifer on the temperature of the overlying material can be illustrated graphically by a series of theoretical isotherm and flow lines. The illustration in Figure 3 shows the heat flow through a wall as affected by the presence of an internal projecting rib (Ingersoll, Zobel and Ingersoll, 1954). It is assumed that the rib has a high thermal conductivity as compared with the wall so that it is an isothermal surface taking the temperature T_1 of the surface of the wall that it joins. This would be similar to an aquifer in the zone affected by surface temperature fluctuations where during the cold and hot months the aquifer maintains a temperature that is the same as the temperature at some greater depth (Cartwright, 1968a). This temperature remains stable although it is situated in the zone affected by the sinusoidal annual air wave.

In summer, the heat flows from high temperature T_2, which repre-

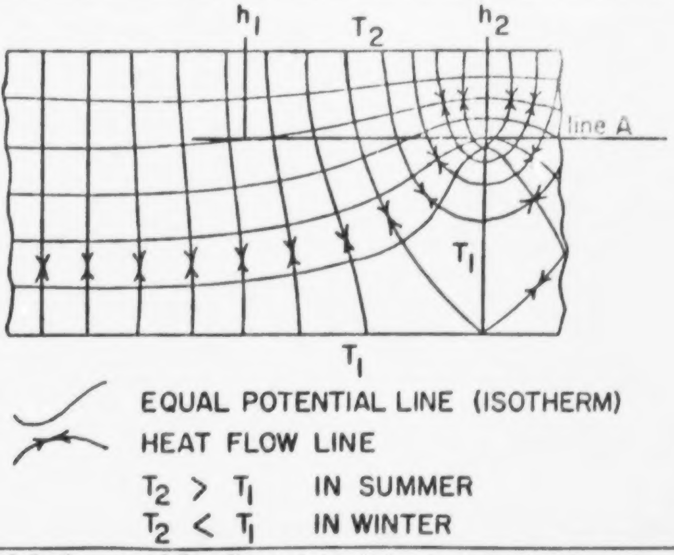


Figure 5 Isotherm and flow lines of steady heat conduction through a soil (modified from Ingersoll, Zobel and Ingersoll, 1954)

sents the earth's surface, to low temperature T_1, at which the aquifer is present. If temperature readings are made in two holes (h_1 and h_2) at a certain depth, the temperature at h_2 will be lower than the temperature at h_1 as indicated by different isotherms with which the bottoms of the holes come into contact. Hence a negative (low temperature) anomaly will be present over the shallow aquifer due to its behavior as a heat sink. In winter, temperature conditions are reversed and a positive (high temperature) anomaly will be present over the aquifer due to its behavior as a heat source.

The size of the anomalies is dependent upon the thermal properties of the overburden, temperature difference between the surface and aquifer and the depth of burial of the aquifer. An estimated maximum temperature anomaly that can be expected during the coldest and warmest months of the year between h_1 and h_2 is about 1.5 C (Cartwright, 1968b).

The surface temperature anomaly decreases as the depth of the aquifer interface increases. Several authors have indicated that 20 meters (65 feet) is the maximum depth of an aquifer that can produce a temperature anomaly detectable by thermal methods.

Description of the Method

In order to detect the temperature anomaly, the effects of diurnal temperature fluctuations should be eliminated. In general, diurnal fluctuations are very small at depths of two feet or greater and they probably occur rather slowly. Thus, it is possible to eliminate most of the diurnal temperature fluctuations by taking a large number of readings at an instant in time and at a depth of two feet or greater. For large surveys that may need a considerable amount of time to be completed, it is better to set up a base station to measure temperature drift. It is sometimes advisable to make at least two time-separated sets of temperature measurements. In practice, an interval of 10 days to two weeks between readings is convenient and reliable (Birman, 1969).

The instrument for thermal surveys consists of a precisely calibrated thermistor at the end of an aluminum rod which can be driven into the ground and a transistor-amplified bridge circuit.

Applications of the Thermal Method to Hydrogeologic Problems

In ground-water monitoring studies, the thermal methods can furnish information which might be unattainable by other geophysical methods. For example, thermal methods are unique in furnishing information concerning the recharge and discharge zones of shallow aquifer systems whereas neither seismic, gravity, magnetic nor electrical methods can supply such information. This information is extremely useful to properly locate a system of monitoring wells.

Locating Shallow Buried Aquifers

Shallow buried aquifers which often can be mapped accurately by the electrical resistivity method can also be determined by the thermal method.

The thermal method has been used extensively to locate shallow buried aquifers by Cartwright (1968a). The thermal surveys were conducted by Cartwright over a 4.5 to 6.0 meters (10 to 20 feet) deep sand and gravel aquifer underlain by Illinoisan glacial till and overlain by a fine-grained Holocene alluvium near Niantic in central Illinois. The aquifer receives recharge from the glacial highland to the northeast and discharges into the Sangamon River to the southwest. The soil temperatures were measured at a depth of 45 cm (18") below the land surface during summer.

The area has been one of extensive resistivity surveys (Buhle, 1953; Emory, 1942). The temperature and apparent resistivity maps were very similar but not identical. Drilling data showed a close correlation between the geology and the temperature anomaly. It showed that the thermal variation related to the buried aquifer reaches -1.5 C in summer, 1 C in winter. The buried aquifer acts as theoretically expected: a heat sink in summer and a heat source in winter.

Smith (1974) conducted a thermal probe survey in an attempt to locate a buried glacial valley. The survey was unable to distinguish a summer low temperature anomaly, indicative of a buried valley. He concluded that the failure of the thermal method was due to the depth of burial of the aquifer. The depth of the buried valley under the till overburden was 27 to 46 meters (90 to 150 feet), which was too deep to contact the summer segment of the annual wave; therefore, a negative anomaly was not produced.

Delineating Ground Water Flow Direction

Cartwright (1974) has also studied the effect of the horizontal flow of ground water on soil temperatures. Horizontal movement of ground water absorbs and redistributes a portion of the rising geothermal heat from the hot interior of the earth, the result being higher soil temperatures in the direction of the ground-water flow. In one study, the soil temperatures increase to the southwest near the area where the water is discharged into the Sangamon River. Soil temperatures increased from 13.9 C near the recharge area to the northeast to 16.3 C seven kilometers southwest near the discharge point, giving a horizontal variation of 0.3×10^{-3} C per meter. Cartwright (1971) estimated that the heat absorbed and redistributed by the aquifer is about 10 percent of the geothermal heat flux.

Delineating Recharge and Discharge Zones

Delineation of the recharge and discharge zones is useful for ground-water monitoring studies. The soil temperatures measured at shallow depths may be used to delineate the recharge and discharge zone for shallow aquifer systems. The vertical movement of ground water in shallow aquifers has also been shown to affect soil temperatures (Cartwright, 1974). Warm water enters the recharge zone, moves downward, and thus warms the soil; as the water moves horizontally, it comes into equilibrium with the general thermal regime; as the water moves upward toward the point of discharge, the soil is cooled. In winter, the water entering the system cools the soil, and water at the discharge point warms it (Cartwright, 1974). The temperature profiles at a road cut near Champaign, Illinois, as shown in Figure 4 were determined by Cartwright (1974) by measuring soil temperatures at a depth of 1 m (3 feet). The site is located near the crest of the Champaign moraine. The soil temperature profile shows a 1-1/2 - 2 C decrease in the summer and a 2 - 3 increase in the winter from the general land surface to the road ditch where ground water seeps. Cartwright found a good agreement between field and theoretical studies and he has concluded that the recharge regions are not generally reflected in the soil temperatures because of the intermittent nature of the recharge events.

Delineating Zones of Leakage Through Earthen Dam Embankments

Birman (1969) stated that thermal technique has been successfully used as a monitoring system for leakage through a dam since 1965. The technique has clearly delineated zones of leakage and has responded to time variations in the amount of leakage. Turner (1977) has conducted a thermal survey of the Wolf Creek Dam located over karstic terrain in Kentucky and he was able to locate zones of solution channels and fractures containing ground water moving at relatively high speeds.

Delineating Zones of Solution Channels, Major Fractures and Joint Systems in Carbonate Rocks

The joints, solution channels and other openings in carbonate rocks generally form a network of interconnected openings that are filled with water moving relatively rapidly. The location of these secondary openings is important for ground water development as well as waste disposal and contaminant studies. Kremar and Masin (1970) have successfully used thermal surveys in Czechoslovakia to find shallow water-bearing cavities and faults. Schneider (1964) has studied well temperatures as a means of tracing ground water movement through carbonate rocks in Israel.

Delineation of Ground-water Contamination

The thermal method has not been extensively used for direct determination of ground-water contamination. The only application that is known to the authors is that by Cartwright and McComas (1968). They have undertaken soil temperature surveys in the vicinity of sanitary landfills in northeastern Illinois to determine if the thermal method can be used to detect and trace the leachate generated from buried refuse. The method indicated the presence of a halo of higher temperatures around the landfill and indicated areas of surface recharge but it could not detect the movement of leachate from the fill. They concluded that the thermal method can directly determine the movement of leachate only when it moves relatively rapidly.

Discussion

There are three basic limitations to the thermal method. First, soil difference, vegetation, and ice in the soil can affect the soil temperatures to the extent that an anomaly will not be produced. Cartwright (1968a) indicated that the effect of vegetation in the summer can be as great as the anomaly produced by an aquifer. Frozen soil in winter months tends to eliminate anomalies, but this effect can be overcome by taking readings at a greater depth (Cartwright, 1968a). Second, only shallow simple aquifer systems can be detected. Third, the information about depth and characteristics of an aquifer by thermal methods is not very reliable (Cartwright, 1968a, b). If these limitations are kept in mind, thermal methods can be a very valuable and inexpensive monitoring tool.

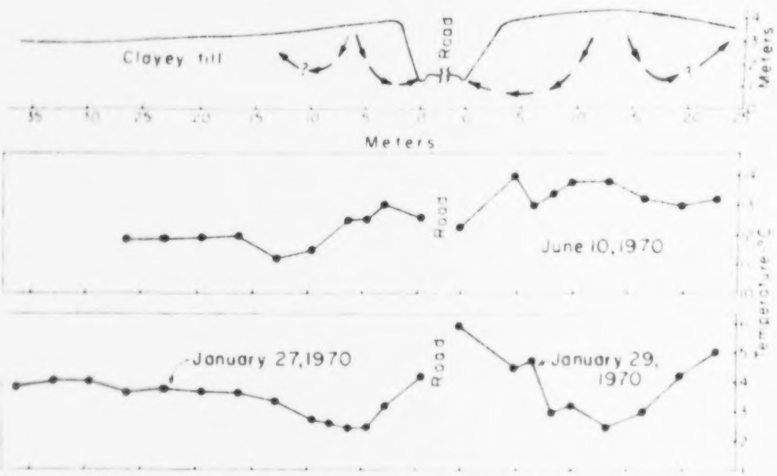


Figure 6. Observed soil temperature profiles (measured at a depth of 1 meter) and geologic cross section at a road cut along the crest of the Champaign moraine, east edge of SW1/4SW1/4 sec. 3 and NW1/4NW1/4 sec. 10, T19N, R8E, Champaign County, Illinois (from Cartwright, 1974).

Summary

This article has been an attempt to review the literature on the application of surface geophysical methods to ground water monitoring. Approximately 60 literature citations have been made relative to those methods most applicable. A review by Zohdy et al., (1974) of the methods as applied to hydrogeologic investigation was the first and best summary made and was cited often. The thermal method has not been used by many, but may have more application for the future; this method was not reviewed by Zohdy et al., in 1974, but we have summarized the literature to date.

It is clear that the only method that can be used for direct measurement of contaminated ground water is the electrical resistivity method. This method also has problems, but the affected water does have a physical property that can be measured and thus makes it an applicable method. It is clear that care must be taken in its application, but given proper planning, the method may be an inexpensive way to monitor ground water quality.

Both the seismic and electrical methods were discussed as to their use toward the development of the geologic framework of a site. They are capable of being used in conjunction with a drilling program to reduce costs in determining the boundary conditions for a ground water flow system at a given site.

Some of the natural and artificial field methods can be used to locate metal objects such as drums or containers used to dispose of waste liquids. Several methods have been used, but are not reported in the literature.

Geophysical methods have a place in a ground-water monitoring program but it is important that the user be aware of all the assumptions made in the development of the particular geophysical method employed.

References

Al'pin, L.M., M.N. Berdichevskii, G.A. Vedrintsev and A.M. Zagarmistr. 1966. Dipole methods for measuring earth conductivity. New York, Consultants Bureau, 302 pp.

Archie, G.E. 1942. The electrical resistivity log as an aid in determining some reservoir characteristics. Amer. Inst. Min. Met. Eng., Tech. Pub. 1422. Petroleum Technology, 8 pp.

Arnow, T., and R. E. Mattick. 1968. Thickness of valley fill in the Jordan Valley east of the Great

Salt Lake, Utah, in Geol. Survey Research, 1968. U.S. Geol. Survey Prof. Paper 600-B, pp. B79-B82.

Berk, W.J., and B.S. Yare. 1977. An integrated approach to delineating contaminated ground water. Ground Water, v. 15, no. 2, pp. 138-145.

Birman, J.H. 1969. Geothermal exploration for ground water. Bull. Geol. Soc. Amer., v. 80, no. 4, pp. 617-630.

Bredehoeft, J.D., and I.S. Papadopoulos. 1965. Rates of vertical ground water movement estimated from the earth's thermal profile. Water Resources Research, v. 1, no. 2, pp. 325-328.

Brown, R.H., A.A. Konoplyantsev, J. Ineson and V.S. Kovalevsky. 1977. Ground water studies: An international guide for research and practice. Unesco.

Buhle, M.B. 1953. Earth resistivity in ground water studies in Illinois. AIME Trans., v. 34, no. 2, pp. 395-399.

Cartwright, K. 1968a. Temperature prospecting for shallow glacial and alluvial aquifers in Illinois. Ill. Geol. Surv. Circ. 433, pp. 1-41.

Cartwright, K. 1968b. Thermal prospecting for ground water. Water Resources Research, v. 7, no. 2, pp. 395-401.

Cartwright, K. 1970. Ground water discharge in the Illinois basin as suggested by temperature anomalies. Water Resources Research, v. 6, no. 3, pp. 912-918.

Cartwright, K. 1971. Redistribution of geothermal heat by a shallow aquifer. Geol. Soc. Amer. Bull., v. 82, no. 11, pp. 3197-3200.

Cartwright, K. 1974. Tracing shallow ground water systems by soil temperatures. Water Resources Research, v. 10, no. 4, pp. 847-855.

Cartwright, K. and M.R. McComas. 1968. Geophysical surveys in the vicinity of sanitary landfills in northeastern Illinois. Ground Water, v. 6, no. 5, pp. 23-30.

Dobrin, M. B. 1960. Introduction to geophysical prospecting. McGraw-Hill Book Co., New York, N.Y., 446 pp.

Domenico, P.A. and V.V. Palciauskas. 1973. Theoretical analysis of forced convective heat transfer in regional ground water flow. Geol. Soc. Amer. Bull., v. 84, pp. 3803-3814.

Eaton, G. P., and J. S. Watkins. 1967. The use of seismic refraction and gravity methods in hydrogeological investigations. Mining and Ground Water Geophysics 1967, Geol. Survey of Canada, Econ. Geol. Report 26, pp. 544-568.

Emory, K.O. 1942. Preliminary report on electrical earth resistivity survey for Oak and Sangamon Ordance plants near Illiopolis, Illinois. Illinois Geol. Survey, open file report, unpublished.

Everett, L.G. 1980. Ground water monitoring. General Electric Co., Schnectady, N.Y. 440 pp.

Fink, W.B. Jr., and D.B. Aulenbach. 1974. Protracted recharge of treated sewage into sand: Part II: tracing the flow of contaminated ground water with a resistivity survey. Ground Water, v. 12, no. 4, pp. 219-223.

Flathe, H. 1967. Interpretation of geoelectrical resistivity measurements for solving hydrogeological problems. Mining and Ground Water Geophysics 1967, Geol. Survey of Canada, Econ. Geol. Report 26, pp. 580-597.

Griffiths, D.H. 1976. Application of electrical resistivity measurements for the determination of porosity and permeability in sandstones. Geoexploration, v. 14, pp. 207-213.

Hackbarth, D.A. 1971. Field study of subsurface spent sulfite liquor movement using earth resistivity measurements. Ground Water, v. 9, no. 3, pp. 11-16.

Heigold, P.C., R.H. Gilkeson, K. Cartwright and P.C. Reed. 1979. Aquifer transmissivity from surficial electrical methods. Ground Water, v. 17, no. 4, pp. 338-345.

Ingersoll, L.R., O.J. Zobel and A.C. Ingersoll. 1954. Heat conduction with engineering, geological and other applications (rev. ed.). Univ. Wisconsin Press, 325 pp.

Johnson, R. B. 1954. Use of the refraction seismic method for differentiating Pleistocene deposits in the Arcola and Tuscola quadrangles, Illinois. Ill. State Geol. Survey Rept. Inv. 176, 59 pp.

Keller, G.V. 1967. Application of resistivity methods in mineral and ground water exploration programs. Mining and Ground Water Geophysics 1967, Geol. Survey of Canada, Econ. Geol. Report 26, pp. 51-66.

Kelly, W.E. 1976. Geoelectric sounding for delineating ground water contamination. Ground Water, v. 14, no. 1, pp. 6-10.

Kelly, W.E. 1977. Geoelectric sounding for estimating aquifer hydraulic conductivity. Ground Water, v. 15, no. 6, pp. 420-425.

Kirk, K.G. and H. Rauch. 1977. The application of the tri-potential method of resistivity prospecting for ground water exploration and land use planning in karst terrains. In Karst Hydrogeology, ed. by J.S. Tolson and F.L. Doyle. UAH Press, The University of Alabama in Huntsville, Huntsville, Ala. pp. 285-299.

Klefstad, G., L.V.A. Sendlein and R.C. Palmquist. 1975. Limitations of the electrical resistivity method in landfill investigations. Ground Water, v. 13, no. 5, pp. 418-427.

Kosinski, W.K. and W.E. Kelly. 1981. Geoelectric soundings for predicting aquifer properties. Ground Water, v. 19, no. 2, pp. 163-171.

Kremar, B. and J Masin. 1970. Prospecting by the geothermic method. Geophysical Prospecting, v. 18, no. 2, pp. 255-260.

Kunetz, G. 1966. Principles of direct current resistivity prospecting. Berlin, Gebruder Borntraeger, 103 pp.

Lennox, D. H., and V. Carlson. 1967. Integration of geophysical methods for ground water exploration in the prairie provinces, Canada. Mining and Ground Water Geophysics 1967, Geol. Survey of Canada, Econ. Geol. Report 26, pp. 517-535.

Levshin, A. L. 1961. Determination of ground water level by the seismic method. Akad. Nauk. SSSR IZU. Ses. Geofiz., no. 9, pp. 857-870.

Lovering, T.S. and H.D. Goode. 1963. Measuring geothermal gradients in drill holes less than 60 feet deep, East Tintic District, Utah: U.S. Geol. Survey Bull. 1172, pp. 1-48.

Manley, T.R. and E.R. Garton Jr. 1977. The ABC's of finding and delineating caves with apparent resistivity measurements. In Hydrologic Problems in Karst Regions, ed. Dilamarter, R.R. and S.C. Csallany, pp. 92-95.

Merkel, R.H. 1972. The use of resistivity techniques to delineate acid mine drainage in ground water. Ground Water, v. 10, no. 5, pp. 38-42.

Ogilvy, A. A. 1967. Geophysical prospecting for ground water in the Soviet Union. Mining and Ground Water Geophysics 1967, Geol. Survey of Canada, Econ. Geol. Report 26, pp. 536-543.

Page, L.M. 1968. The use of the geoelectric method for investigating geologic and hydrologic conditions in Santa Clara County, California. Jour. Hydrology, v. 7, no. 2, pp. 167-177.

Parsons, M.L. 1970. Ground water thermal regime in a glacial complex. Water Resources Research, v. 6, no. 6, pp. 1701-1720.

Poole, V. L., and P. C. Heigold. 1981. Geophysical assessment of aquifers supplying ground water to eight small communities in Illinois. Ill. Geol. Survey, Environmental Geol. Notes 91, pp. 1-61.

Sayre, A.N. and E.L. Stephenson. 1937. The use of resistivity methods in the location of salt-water bodies in the El Paso, Texas, area. Amer. Geophy. Union Trans., v. 18, pp. 393-398.

Schneider, R. 1962. An application of thermometry to the study of ground water. U.S. Geol. Surv. Water Supply Pap. 1544-B., pp. B1-B16.

Schneider, R. 1964. Relation of temperature distribution to ground water movement in carbonate rocks of central Israel. Bull. Geol. Soc. Amer., v. 75, pp. 209-216.

Sendlein, L. V. A., and R. C. Palmquist. 1975. A topographic-hydrogeologic model for solid waste landfill siting. Ground Water, v. 13, no. 3, pp. 260-268.

Smith, E.M. 1974. Exploration for a buried valley by resistivity and thermal probe surveys. Ground Water, v. 12, no. 2, pp. 78-83.

Stallman, R.W. 1963. Computation of ground water velocity from temperature data. Methods of Collecting and Interpreting Ground Water Data, ed. R. Bentall, U.S. Geol. Surv. Water-Supply Paper 1544-H, pp. H36-H46.

Stallman, R.W. 1965. Steady one-dimensional fluid flow in a semi-infinite porous medium with semi-solid surface temperature. J. Geophys. Res., v. 70, no. 12, pp. 2821-2827.

Staub, W.P. 1969. Seismic refraction, a technique for subsurface investigation in Iowa. Unpub. Ph.D. thesis, Iowa State University, Ames, Iowa.

Stollar, R.L. and P. Roux. 1975. Earth resistivity surveys—a method for defining ground water contamination. Ground Water, v. 13, no. 2, pp. 145-150.

Stone, R. 1971. Hydrogeology of the Nishnabotna river basin. Unpub. Ph.D. thesis, Iowa State University, Ames, Iowa.

Swartz, J.H. 1937. Resistivity studies of some salt water boundaries in the Hawaiian Islands. Amer. Geophy. Union Trans., 18th Annual Meeting, Part 2, pp. 387-393.

Turner, W.M. 1977. Use of thermonics in foundation studies in karstic terrain. In Hydrologic problems in Karst Regions, ed. Dilamarter, R.R. and S.C. Csallany, pp. 459-462.

Van Dam, J.C. 1976. Possibilities and limitations of the resistivity method of geoelectrical prospecting in the solution of geohydrological problems. Geoexploration, v. 14, pp. 179-193.

Van Dam, J.C. and J.J. Meulenkamp. 1967. Some results of the geoelectrical resistivity method in ground water investigations in the Netherlands. Geoexploration, v. 14, pp. 179-193.

Van Nostrand, R.G. and K.L. Cook. 1966. Interpretation of resistivity data. U.S. Geol. Survey Prof. Paper 499, 310 pp.

Warner, D.L. 1969. Preliminary field studies using earth resistivity measurements for delineating zones of contaminated ground water. Ground Water, v. 7, no. 1, pp. 9-16.

Zohdy, A. A. R. 1964. Earth resistivity and seismic refraction investigations in Santa Clara County, California. Unpub. Ph.D. thesis, Stanford University, 132 pp.

Zohdy, A. A. R. 1965. Geoelectrical and seismic refraction investigations near San Jose, California. Ground Water, v. 3, no. 3, pp. 41-48.

Zohdy, A.A.R. 1969. The use of Schlumberger and equatorial soundings in ground water investigations near El Paso, Texas. Geophysics, v. 34, no. 5, pp. 713-728.

Zohdy, A.A.R. 1974. Use of Dar Zarrouk curves in the interpretation of vertical electrical sounding data. U.S. Geol. Surv. Bull., 1313-D, pp. 1-41.

Zohdy, A.A.R. and R.J. Bisdorf. 1975. Computer programs for forward calculation and automatic inversion of Wenner sounding curves. USGS PD 75-021.

Zohdy, A. A. R., G. P. Eaton, and D. R. Mabey. 1974. Application of surface geophysics to ground water investigations. Techniques of Water Resources Investigations, U.S. Geol. Survey, Bk. 2, Chap. D1, pp. 1-116.

Zohdy, A.A.R. and D.B. Jackson. 1969. Application of deep electrical soundings for ground water exploration in Hawaii. Geophysics, v. 34, no. 4, pp. 584-600.

SIGNIFICANCE TESTING FOR IMPACT EVALUATION

By Frank A. Rovers and
Edward A. McBean

Introduction

In an attempt to establish the significance of changes in environmental quality, widespread use is being made of the Student's t-test. For example, in the investigations of statistical test procedures which could be useful in interpreting ground-water monitoring data, the Environmental Protection Agency has specified the t-test with regard to the Resource Conservation and Recovery Act (RCRA). As originally proposed, analyses of ground-water quality were to be compared to the background quality established for each facility using the single tailed Student's t-test at the 95 percent confidence level.

After considering a number of detracting comments about the Student's t-test, the Agency reproposed the statistical test on September 19, 1979, (44 Federal Register (FR) 54323-54324) and specified the use of the Mann-Whitney U-test at the 95 percent confidence level. However, commentators on the reproposal generally preferred the Student's t-test over the Mann-Whitney U-test and the Agency again specified the Student's t-test, but at the 99 percent level of significance instead of the originally proposed 95 percent level (45 FR 33194-33195). Current regulations being enforced at the time of writing specify this testing procedure.

Although the Student's t-test is highly useful in discriminating between sets of data, there is an underlying assumption regarding its variance that precludes the use of this test for significance testing in some situations. For example, the impact of an effective remedial measure on ground-water quality will adjust both the mean value of concentration and the variance of the concentrations. The development of a common variance for use in the t-test may be inappropriate and the result will be an inappropriate subsequent utilization of the t-test. To a large extent, the Mann-Whitney U-test circumvents this problem, but, being a non-parametric test, it does not have the discrimination power of the t-test. Therefore, it is the intent of this paper to indicate that a data transformation may be necessary to satisfy the t-test requirements and thus allow it to be employed. Case study examples are used to demonstrate the procedure.

Significance Testing

When a finding which is very unlikely to have arisen by chance is obtained—such as the difference between background ground-water quality and contaminated ground water—the result is referred to as being statistically significant. In other words, the finding is difficult to ascribe to chance and the difference must, in all common sense, be accepted as a real difference. Since this judgment is based on probability and falls short of absolute certainty, the degree of confidence in the difference should be described as "probably significant" (one chance in twenty or a probability of 0.05), "significant" (one chance in one hundred or a probability of 0.01) or "highly significant" (one chance in 1,000 or a probability of 0.001), depending on the probability level associated with the judgment.

Because the judgment is based on probability, there are two risks in discrimination testing:

- deciding that a real difference exists, when in fact there is none
- deciding that no difference exists, when in fact there is a difference.

As a means of reflecting the probability, a series of discrimination testing procedures have evolved, with the Student's t-test being perhaps the most commonly used.

Student himself actually considered only the problem of testing whether a set of observations with mean $\bar{x}$ could have come from a population distribution with mean μ. It was Fisher (1925) who pointed out that the t-test could be applied to a number of other problems, in particular the testing of whether the means of two populations are the same.

Mathematically, the procedure allows the testing of whether the means from two sets of measurements, say X (where elements of X are x_i where $i = 1, \ldots m$) and Y (where elements of Y are y_j where j ′ 1, ...n), are the same. Assuming that X and Y are normally distributed with the same variance but that their population means μ_x and μ_y may be different, then the difference between the means $\bar{x} - \bar{y}$ will be normally distributed with mean $(\mu_x - \mu_y)$ and variance $\sigma^2 \left(\frac{1}{m} + \frac{1}{n}\right)$

Then

$$t = \frac{|\bar{x} - \bar{y}|}{s\sqrt{\frac{1}{m} + \frac{1}{n}}} \qquad (1)$$

where $|\ |$ denotes the absolute value sign and 's' represents the standard deviation, will follow a t-distribution with $m + n - 2$ degrees of freedom. Noteworthy points regarding the above include:

- The assumption that distributions of x and y have the same variance is essential to the argument.

- The variance $\sigma^2 (\frac{1}{m} + \frac{1}{n})$ is normally referred to as the common variance. (This will be discussed more fully below.)
- The t-test is thus based on the assumption that this underlying distribution is normal or gaussian. When the number of observations is large, the central limit theorem (e.g. see Benjamin and Cornell, 1970) states that the sample mean will be approximately normally distributed regardless of the underlying distribution. The result is that the t-distribution is remarkably insensitive to small or moderate departures from normality. Unfortunately, both of these assumptions may be violated in ground-water monitoring results, since the data can be skewed [see Rovers and McBean (1981) for further details] and the number of observations is often small due to the expense and difficulty of sampling.

A result of fundamental importance in the theory of significance testing is that the variance of the sum or difference of two independent random variables is equal to the sum of their variances. The variance of the average of n samples drawn from a population whose standard deviation is equal to σ is given by σ^2/n. Therefore, a sample of n items drawn from a population whose standard deviation is σ_1 and a sample of m items drawn from a population whose standard deviation is σ_2, will have a variance for the distribution of the difference of sample means, $\bar{x}_1 - \bar{x}_2$, given by

$$\mathrm{Var}(\bar{x}_1 - \bar{x}_2) = \frac{\sigma_1^2}{n} + \frac{\sigma_2^2}{m} \qquad (2)$$

Since departures greater than two standard deviations occur relatively rarely (of the order of once in twenty trials on the basis of random sampling), the above provides a means of testing the statistical significance of the difference between sample means. The standard deviation for the distribution of sampling means is usually referred to as the standard error of the difference.

Before examining the implications of the above in greater detail, it is first essential in analyses of most ground-water monitoring data problems to remove statistical bias in the estimate of the standard deviation or variance.

Statistical Bias

While the mean value of a sample of n items is an unbiased estimate of the mean value in the population from which the sample is drawn, the standard deviation is biased, tending to underestimate the population value. This bias is especially marked in the small sample sets typical of ground water monitoring data.

It is relatively easy to show [e.g. see Miller and Freund (1965)] that the expected value of the variance in a sample of n items, s^2, is related to the population variance, σ^2, as

$$E[s^2] = (\frac{n-1}{n})\sigma^2 \qquad (3)$$

where the symbol $E[s^2]$ denotes the expected value for the variance of the sample. The factor $(\frac{n-1}{n})$ is Bessel's correction. Therefore, the best estimate of the population variance, $\hat{\sigma}^2$, obtained from the sample variance, s^2, is

$$\hat{\sigma}^2 = (\frac{n}{n-1})s^2 \qquad (4)$$

Note that $\hat{\sigma}$ is always greater than s with the difference between the two values becoming very small as the sample size n becomes large.

Significance Testing on Variance Estimates

A key to the validity of the t test is the assumption that the distributions of x and y have the same variance. Therefore, before using the t test to investigate the difference between the sample means, a prior test should be done to investigate whether the sample variances are sufficiently alike to warrant the assumption that they are independent estimates of the same population variance.

The significance of the difference between sample variances can be tested using the variance ratio test, sometimes referred to as Snecedor's F test. The variance ratio is defined as:

$$F = \frac{\text{greater estimate of the variance of the population}}{\text{lesser estimate of the variance of the population}} \qquad (5)$$

where the variance estimates are determined using Bessel's correction to the sample variances. It follows that the larger F is, the less likely it is

Table 1

5% Level of Variance Ratio Associated with Equation 5

Number of degrees of freedom in the greater variance estimate

Number of degrees of freedom in lesser variance estimate	1	2	3	4	5	10	20	∞
1	161	200	216	225	230	242	248	254
2	18.5	19.0	19.2	19.2	19.3	19.4	19.4	19.5
3	10.1	9.6	9.3	9.1	9.0	8.8	8.7	8.5
4	7.7	6.9	6.6	6.4	6.3	6.0	5.8	5.6
5	6.6	5.8	5.4	5.2	5.0	4.7	4.6	4.4
10	5.0	4.1	3.7	3.5	3.3	3.0	2.8	2.5
20	4.3	3.5	3.1	2.9	2.7	2.3	2.1	1.8
∞	3.8	3.0	2.6	2.4	2.2	1.8	1.6	1.0

Ref: Fisher and Yates

Table 2

1% Level of Variance Ratio Associated with Equation 5

Number of degrees of freedom in the greater variance estimate

Number of degrees of freedom in lesser variance estimate	1	2	3	4	5	10	20	∞
1	4100	5000	5400	5600	5800	6000	6200	6400
2	98.0	99.0	99.0	99.0	99.0	99.0	99.0	99.0
3	34.0	31.0	29.0	29.0	28.0	27.0	27.0	26.0
4	21.0	18.0	17.0	16.0	16.0	15.0	14.0	13.0
5	16.0	13.0	12.0	11.0	11.0	10.0	9.6	9.0
10	10.0	7.6	6.6	6.0	5.6	4.8	4.4	3.9
20	8.1	5.8	4.9	4.4	4.1	3.4	2.9	2.4
∞	6.6	4.6	3.8	3.3	3.0	2.3	1.9	1.0

Ref: Fisher and Yates

that the Null Hypothesis (that two samples are drawn from the same population) is a valid one. However, with small sample sizes (m and n small), it is possible that F could be different from unity. Large samples, if the Null Hypothesis is true, should give a value of F which differs little from unity. Tables 1 and 2 show the value of F which will be exceeded with a given degree of probability for various sample sizes. (Note that the number of degrees of freedom for a sample of n items is equal to n − 1.)

When m and n are large, an alternative procedure is to test the significance of a difference in standard deviations using the standard error of difference method and the fact that the variance for the distribution of sample standard deviations is $\sigma^2/2n$ (note: it is the standard deviation and not the mean) and the fact that the variance of a difference is equal to the sum of the variances.

In the event that there is a significant difference between the variances, the assumptions of the t-test are violated and erroneous conclusions may be determined. At least two alternatives are possible:

- Utilize a different discrimination test such as the Mann-Whitney U test (e.g. see Willemain, 1980), a non-parametric test.
- Transform the data (e.g. by logarithms or square root) and utilize the Variance Ratio test on the transformed data.

Very briefly, the Mann-Whitney U test involves a joint arrangement of the data points in accordance with their size, retaining the sample identity of each observation. Ranks are then assigned to the observations. If two or more observations are tied in rank, each of the observations is assigned the mean of the ranks they jointly occupy.

Denoting the respective sample sizes by n and m and the sum of the ranks occupied by the first sample by R_1, it can be shown that the mean and the variance of the sampling distribution of the statistic

$$U = nm + \frac{n(n+1)}{2} - R_1 \qquad (6)$$

are given by

$$\mu_u = \frac{nm}{2} \qquad (7)$$

and

$$\sigma_u^2 = \frac{nm(n+m+1)}{12} \qquad (8)$$

If both n and m are greater than 8, the distribution of the U statistic can be approximated closely by a normal distribution and hence, the test can be based on the statistic

$$z = \frac{U - \mu_u}{\sigma_u}$$

With knowledge of z, conclusions can then be made as to whether the Null Hypothesis of identical populations can or cannot be rejected.

Case Study 1

During the monitoring of a landfill site being used for methane gas recovery [see Crutcher et. al. (1981) for further details], it was necessary to identify a monitoring probe that was not affected by pumping for gas extraction. If a measure of ambient conditions could be identified, then comparisons with other results obtained at other monitoring probes would allow the calculation of the difference in concentrations (drawdown from ambient or background). The impact of the pumping could then be characterized.

To test the validity of the assumption that the monitoring point is not affected by pumping, the monitoring results can be examined to determine if there is a significant difference in the methane concentrations with different pumping rates. Consider the results obtained during two pumping tests, summary details of which are indicated in Table 3. Using Bessel's correction [Eqn. (4)], the variance estimates are $\hat{\sigma}_x^2 = 41.7$ and $\hat{\sigma}_y^2 = 37.7$ which gives an F-ratio as

$$F = \frac{41.7}{37.7} = 1.1 \qquad (9)$$

From Table 1 to 7 degrees of freedom for the greater variance value and 12 degrees of freedom for the lesser variance, it is found that the difference between the variances is not significant, thereby allowing the t-test to be utilized.

Given the above, a pooled estimate of the assumed common variance is

$$\hat{\sigma}^2 = \frac{n\,\hat{\sigma}_x^2 + m\,\hat{\sigma}_y^2}{n+m} = 39.2 \qquad (10)$$

The standard error of difference of the two means is

$$\hat{\sigma}_m = \sqrt{\hat{\sigma}^2 \left(\frac{1}{n} + \frac{1}{m}\right)} \qquad (11)$$

$$= \sqrt{39.2 \left(\frac{1}{8} + \frac{1}{13}\right)} = 2.81 \qquad (12)$$

The Student t is then calculated as

$$t = \frac{\text{Difference of Means}}{\text{Standard Error of Difference}} \qquad (13)$$

$$= \frac{50.8 - 44.3}{2.81} = 2.31$$

The number of degrees of freedom for Student's t = 8 + 13 − 2 = 19.

Referring the calculated value of t to tables of Student's t (e.g. see Benjamin and Cornell, 1970), the observed difference could have arisen by chance in one trial in 50 so that the difference is significant. The t-test thus allows the conclusion that the measurements at the monitoring probe are affected by pumping rate.

Case Study 2

Measurements made upstream (i.e. background) and downstream of a potential contaminant source are listed in Table 4. A pairwise comparison of individual values indicates that the contaminant source is contributing. However, calculating a common variance provides

$$\hat{\sigma}^2 = \frac{10(15.6) + 10(69.4)}{10 + 10} = 42.5 \qquad (14)$$

and a standard error of the differ-

Table 3

Statistics of Methane Concentrations as a Function of Pumping Rate

Set	Gas Pumping Rate	No. of Samples	CH$_4$ Concentration Statistics Mean (% by volume)	Standard Deviation (% by volume)
	(cfm)		x	s_x
X	90	8	44.3	6.04
Y	50	13	50.8	5.90

ence of the two means as

$$\hat{\sigma}_m = \sqrt{42.5\left(\frac{1}{10} + \frac{1}{10}\right)} = 2.92 \qquad (15)$$

The Student t becomes

$$t = \frac{6.14 - 1.33}{2.92} = 1.64 \qquad (16)$$

For 18 degrees of freedom, the significance of the difference is not established, contrary to the conclusion one reaches with a pairwise comparison.

Nevertheless, an examination of ratio of the variances shows that the F value is

$$F = \frac{69.4}{15.6} = 4.45 \qquad (17)$$

A check of Table 1 indicates that the difference between the variance is probably significant. Therefore, using a square-root transform (other possibilities also could be used), the data in Table 4 become those of Table 5. The value of F for the transformed data is

$$F = \frac{1.58}{1.08} = 1.46 \qquad (18)$$

indicating there is no longer a significant difference between the variances. The common variance can now be calculated as

$$\hat{\sigma}^2 = \frac{10(1.08) + 10(1.58)}{10 + 10} = 1.33 \qquad (19)$$

and the standard error of the difference between the two means as

$$\hat{\sigma}_m = \sqrt{1.33\left(\frac{1}{10} + \frac{1}{10}\right)} = .516 \qquad (20)$$

The Student t becomes

$$t = \frac{2.20 - .6}{.516} = 2.97 \qquad (21)$$

and the significance of the difference between the means is established.

As an alternative to the above, the ranked data for the Mann-Whitney U-test is contained in Table 6. Defining R_2 as the sum of the ranks occupied by the downstream sample, for the data then

$$n = 10 \quad \text{and} \quad m = 10$$
$$R_2 = 64 \qquad (22)$$

and

$$U = 91$$
$$\mu_u = 50$$
$$\sigma_u^2 = 175 \qquad (23)$$

so

$$z = \frac{91 - 50}{\sqrt{175}} = 3.09 \qquad (24)$$

From normal probability tables, with a value of $z = 3.09$, it is apparent that the null hypothesis can be rejected (and inference can be made in this situation the two samples do not come from the same population).

In this example, the preponderance of the rankings seen from Column 2 of Table 6 makes the results rather intuitively obvious, but this will not always be the situation.

Table 4

Concentrations, Upstream and Downstream of a Contaminant

Concentrations (μg/ℓ)

	Upstream (Background) Location	Downstream Location
	(U)	(D)
1	0.29	4.3
2	12.0	28.0
3	.32	6.1
4	.049	2.66
5	.14	3.0
6	.158	6.42
7	.07	5.74
8	.07	1.40
9	.07	1.49
10	.14	2.3
Mean	1.33	6.14
Standard Deviation	3.75	7.90
Unbiased Variance Estimate	15.6	69.4

Table 5

Square-Root Transformed Data of Concentration

	Upstream (Background) Location	Downstream Location
	(U)	(D)
1	.54	2.07
2	3.46	5.29
3	.57	2.47
4	.22	1.63
5	.37	1.73
6	.40	2.53
7	.26	2.40
8	.26	1.18
9	.26	1.22
10	.37	1.52
Mean	.67	2.20
Standard Deviation	.99	1.19
Unbiased Variance Estimate	1.08	1.58

Conclusions

When the assumptions of the t-test are properly met, the student's t-test and the Mann-Whitney U-test provide the same results. Nevertheless, being a non-parametric technique, the Mann-Whitney U-test loses information when the real variabilities are replaced by ranking numerals. In the case of large samples this loss of information by the Mann-Whitney test is unimportant, but for small samples—the common situation when dealing with ground-water monitoring—the discrimination capabilities of the Mann-Whitney U-test are not as great.

Transformation of the data may be required for the t-test to resolve the problem of different variances, an assumption implicit within the t-test.

Using the Null Hypothesis, if the results prove non-significant, this is equivalent to a verdict of "not proven." Further analyses are still appropriate given further evidence.

Biographical Sketches

Frank A. Rovers is a senior partner in the firm Conestoga-Rovers and Associates; Edward A. McBean is a professor in the Department of Civil Engineering, University of Waterloo and a technical advisor to Conestoga-Rovers and Associates.

References

Benjamin, J. R., and Cornell, C. A., 1970. *Probability, Statistics, and Decision for Engineers*. McGraw-Hill Book Co., New York.

Crutcher, A., Rovers, F. A., and McBean, E. A., 1981. "Methane Gas Utilization from a Landfill Site." *ASCE-Journal of the Environmental Engineering Division*, forthcoming.

Fisher, R., Yates, D., *Statistical Tables for Biological, Agricultural and Medical Research*. Oliver and Boyd Ltd., Edinburgh.

Miller, I. and Freund, J. E., 1965. *Probability and Statistics for Engineers*. Prentice-Hall Inc., Englewood Cliffs, New Jersey.

Rovers, F. A., and McBean, E. A., 1981. "An Assessment of Alternatives of Analysis of Monitoring Program Data for Solid and Hazardous Waste Disposal Sites," submitted to *Ground Water Monitoring Review*.

Willemain, T. R., 1980. *Statistical Methods for Planners*. The MIT Press, Cambridge, Massachusetts.

Table 6

Ranked Data for Mann-Whitney U-Test

	Location at which Measurement Taken	Concentration (μg/ℓ)	Rank
1	D	28.0	1
2	U	12.0	2
3	D	6.42	3
4	D	6.1	4
5	D	5.74	5
6	D	4.3	6
7	D	3.0	7
8	D	2.66	8
9	D	2.3	9
10	D	1.49	10
11	D	1.40	11
12	U	.32	12
13	U	.29	13
14	U	.158	14
15	U	.14	15
16	U	.14	16
17	U	.07	18
18	U	.07	18
19	U	.07	18
20	U	.049	20

U.S. ARMY INSTALLATION RESTORATION DATA MANAGEMENT

By Allen W. Shatto, U.S. Army Toxic and Hazardous Materials Agency

Introduction

For over 30 years the military departments of DOD have been engaged in a wide variety of operations with military and industrial chemicals. In some cases the migration of these materials through surface and subsurface waters beyond the borders of the military property has resulted in litigation. In other cases facilities have become available for other public or private uses provided they could be restored to a safe condition. These situations coupled with the enactment of broad-based legislation on pollution control have led the DOD to implement an environmental quality enhancement program to abate pollution from past operations which have an immediate impact on public health and to clean up residual pollutants at those installations which may be released in the future.

The DOD, in July 1976, designated the Department of the Army as the lead agency for the development of technology and standards for the pollution control program as it relates to chemical (military, hazardous and explosive), biological and radiological pollutants resulting from past operations. The basic approach was that of classical problem-solving: define the current situation, identify alternative courses of action to reach the desired state, and perform a comparative analysis of the alternatives. In the Army Installation Restoration Program, the current situation is defined by environmental surveys of chemical, biological and radiological contamination of the soil, surface water, ground water, biological tissue and man-made structures. The allowable condition is usually defined by existing standards and regulations promulgated by the EPA or state environmental agencies. However, for military unique compounds, the U.S. Army Surgeon General is responsible for developing the toxicity data base and proposing standards where they may not otherwise exist. Alternative courses of action to contain and/or treat the contamination are identified by a decontamination engineering program. Data are generated from each of these phases and stored in a data base on the Univac 1108 computer at the Edgewood Area of Aberdeen Proving Ground, Maryland. When prevailing standards exist in EPA or other agency data bases, computer terminal links are made to those data bases rather than incurring duplicate effort in maintaining current entries. The comparative analyses draw the necessary information from all the data bases and, through an iterative process of matching removal efficiencies to contaminant concentrations and standards, produce a decision matrix of cost versus residual contamination for the decision-makers' final determination.

The need to computerize the data is twofold. First, the volume and complexity of the data require the application of computer technology to rapidly locate and retrieve specific pieces of information from the estimated two million data records which may result from a full scale project. Secondly, the ability of the computer to graphically present spatially distributed data in a comprehensible form is much superior to that of a similar manual system, particularly from a cost-effectiveness standpoint. (A sometimes hidden benefit of computerizing these operations is the additional number of retrievals and presentations that are possible within project schedules which simply could not be done if they had to be constructed manually.)

Technical Approach

The three basic components of the Data Management Program are distributed processing, centralized data base and hardware software standardization. Distributed processing is accomplished by locating mini-computer terminals on site during the survey operation. These mini-computers can do data entry, checking and editing functions as well as limited statistical analyses without incurring communication costs or occupying premium Univac 1108 main computer time. The centralized data base maintains control of formats and allowable entries, provides the "same sheet of music" to all performers, and provides computing power and mass storage for the total data base. Hardware and software standardization minimizes duplication of effort since a nucleus of computer programmers can write the required programs once and distribute "canned" copies on magnetic tape to the various users. Furthermore, standardization allows innovation by one field unit to the shared by all users without translation to other computing languages or operating systems. Similarly, any problems solved once are indeed solved "for all." Finally, standardization in the Army program reduces follow-on costs of new installations since the equipment and all necessary software are moved as a known operating configuration from site to site as projects are completed and others begun.

The specific automatic data processing equipment used on this program are the Univac 1108 computer with seven 150 megabyte mass-storage discs, a Calcomp 936 plotter and numerous other peripheral devices of only incidental importance. Our mini-computer terminals are Tektronix 4050 series units with expandable memory from 8 to 32 thousand bytes depending on the site needs. We have found that the field units should have a minimum configuration including a hard copy unit which takes photostatic copies of screen maps in about five seconds, a small (8-1/2" x 14") bed plotter which makes felt-pen maps in 15 minutes, and a printer for tabular output.

Whenever possible we have used commercially available software and "tailored" it to our application. Most of the two-dimensional mapping done to date has used the Calcomp General Purpose Contour Package. Current efforts in three-dimensional work makes use of Kansas Geological Survey Surface II and Dynamic Graphics Surface Display Library. The application software defining Army contamination data for these packages was written by programmers from Chesapeake Software under Army contract.

For a brief look at the capabilities of this tailored software, a stepwise sequence through a hypothetical project will be presented. At the initiation of the project, a project officer is appointed to be responsible for that installation's survey and subsequent decontamination. After a review of available information about the site, the project officer marks a U.S. Geological Survey map with the natural and manmade features which are likely to be significant in the project. The marked-up map is provided to the programmers who use an electronic tablet with an optical crosshair to digitize (convert lines into x, y coordinates) all significant features to scale.

Next the sampling points are located either by surveying or by plotting on the base map with respect to landmarks. These sampling points may be wells, bores, streams, lakes, buildings or other types of sampling points, but each one is given a unique site identifier (e.g. WELL 76; BORE 102). From this point on all analyses are associated with the unique sampling site identifier and the computer retrieves the map coordinates. A ground-water monitoring program may use existing wells or drill new wells. In the event new wells are drilled, the bore logs at each location are entered into the computer. A bore log profile can be constructed automatically by specifying the end points of a segment and the distance of interest on either side of the segment. The transect end points may be specified by site identifiers or by coordinates. The computer retrieves all data in the specified area and plots the bore logs to scale along the transect and vertically with respect to mean sea level.

After well construction, the water table is allowed to stabilize and depth measurements are taken. The computer will convert depth to elevation using the bore log information and plot water tble elevation contours. As the field drilling work is being completed, the chemical analysis laboratory is being certified.

The Army requires each analytical laboratory to demonstrate competence by producing precision and accuracy data on the analysis of known concentrations of each analyte. When this data is reviewed and accepted the laboratory is added to the data base of certified laboratories for that method. Without certification any chemical analysis data submitted to the data management system are rejected.

Now samples are taken from the wells, labeled with sample control numbers and sent to the laboratory. The sampling portion of the data record will indicate where the sample was taken, the date, the depth and the sampling technique. The laboratory will perform the analyses and complete the data record. Once these data are transmitted to the central data base and accepted, concentration contour maps can be constructed. In the monitoring configuration, contour maps for consecutive periods can provide a time series "movement" of contamination plumes.

To this point all previous discussion has dealt with historic data. However, to design a containment or treatment system to be effective for 20 years or more requires the best possible predictive estimates of future conditions, that is, simulation. To this end the Army has contracted with Arthur D. Little Inc. to develop a total water regime model.

Various investigators and organizations have modeled parts of the water regime such as surface runoff, streams and aquifers. The Army's approach was to use available models where possible and develop linkages between them to pass output from one to another. Therefore a modular approach was undertaken with an interdisciplinary team of specialists acting as module leaders. A system analyst was responsible for integration and software architecture. The modules reflect the elements of the overall program: water transport (including chemical concentrations), containment, treatment, ecology and cost/benefit analysis. The largest and most complex module was water transport.

The water transport module commences with a projection of meteorological conditions and storm events. The projection is based on 10 years of historical data from the National Weather Service station nearest the modeled site. A Markov projection is made in predicting a monthly precipitation and the data base is searched for the same calendar month most closely approaching the estimate. The hourly precipitation, temperature, wind velocity, relative humidity and so forth, are used from the historical month selected. Precipitation as snow is passed through freeze-melt cycles as temperatures vary. Water from melting snow and from rain are partitioned at the ground surface, into overland flow and infiltration with the proportion of each depending on the use of the land. Overland flow is coupled to the local streams and tributaries via watershed mapping of the surface typography. Infiltration is coupled to the ground-water system either through an unsaturated zone model or directly to the saturated zone. The aquifers of the saturated zone are reconnected with the streams either as stream recharge or sinks. Once the physical system is described, a nodal network is designed to pass variables and a calibration run is made to confirm the model's representation of known data. A "baseline" run is made to predict the future results of contaminant migration under the existing conditions. All containment and treatment options are compared to this baseline.

The containment module has two functions: to alter the flow characteristics of the water system and to estimate costs of the various options. Current options include caps, liners, dams, diversions, barriers and pumping well systems. The user is required to select the location and geometry of each containment option. The model will alter the permeability and direction of flow and provide "first cut" cost estimates for the options selected. The contaminant concentrations may be rerun in the water transport module or passed to the decontamination module.

The decontamination module takes volumetric flow estimates and feeder-stream concentrations of multiple contaminants and searches a library of chemical engineering unit operations, providing preliminary cost estimates. Since multiple contaminants may be addressed, it is quite probable that multiple stages of different unit operations may be required in a process train to be effective in lowering all contaminants to acceptable levels. Furthermore, feed stream concentrations may vary with time depending on the strength, location and mobility of contaminant sources. The user specifies the uptake points from the water transport module and the concentration profiles of the contaminants are determined. (There may or may not be contaminant options tested simultaneously.) The selection process begins with the contaminant most in excess of the allowable standard. Unit operations are selected for the first stage which have some removal efficiency for this contaminant and are not poisoned by one or more of the other contaminants. The removal efficiency for each contaminant is retrieved from a treatability matrix and the effluent stream is compared to allowable standards for all contaminants and, if any contaminant is in excess, a second stage is selected. This process continues until either all standards are met or the user specifies a maximum number of stages. The selection process is repeated for each period of time up to the time when untreated contaminants fall to acceptable levels. The selected processes are costed for both capital and operating cost. The influent and effluent concentrations and the costs are stored along with the containment costs for use in the cost/benefit module.

The cost/benefit module is actually a top-level report generator. A concentration report is generated showing the simulated influent and effluent concentrations for each contaminant during each time period. A cost report shows the annual cash flow for capital and operating costs by fiscal year and may be inflated or discounted by user specification. These reports are turned over to the decision-making individual or group for final determination.

Progress to Date

The Army has collected some 20 million data bits from 14 installation restoration projects and expects to have about 11 more installations active by the end of fiscal year 1981. The automated data base on the Univac 1108 computer is structured and operational. Data entry is automatic from the remote terminal sites. Access and retrieval is being done on a System 2000 Data Base Management System marketed by MRI. Digitizing, bore log profiling and two-dimensional contour plotting are fully operational. The simulation and cost/benefit analysis have been designed and software completed. Proof testing of the ground-water simulation has been completed and the first full scale application as a decision-making tool is under way. Surface-water development has been completed and initial testing is being conducted this year.

Future Plans

Three-dimensional mapping of contamination plumes and geologic structures are programmed in the current fiscal year. The next generation graphics will incorporate the ability to present dynamic maps through moving picture presentations linking the computer to videotape output. Coupling these presentation modes to the simulation capability will bring the environmental community that much closer to being able to use today's data to observe tomorrow's world and make the right decisions in protecting our environment.

Allan W. Shatto is the engineering team leader for Decontamination Technology and Data Management in the U.S. Army Toxic and Hazardous Materials Agency, Aberdeen Proving Ground, Maryland. He has designed and implemented the Army's automatic data collection system for installation restoration environmental surveys. Currently, data is being collected from 15 installations with 12 additional sites scheduled this year. Shatto has a BS degree in chemical engineering from Lehigh University and a MS degree in administration of research and development from George Washington University.

- END -

DATE Filmed
01/ 19 / 83